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Superfund
SITE: Centredale
BREAK: 3-4
OTHER: 206666

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US ARMY CORPS
OF ENGINEERS
New England District

Contract No. DACW33-01-D-0004
Delivery Order No. 01
September 2004

INTERIM-FINAL BASELINE ECOLOGICAL RISK ASSESSMENT

VOLUME I OF II

Centredale Manor Restoration Project Superfund Site North Providence, Rhode Island

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SDMS DocID 000206666

BASELINE ECOLOGICAL RISK ASSESSMENT – INTERIM FINAL

CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

Submitted to:

Department of the Army
U.S. Army Corps of Engineers
New England Division

Contract No. DACW33-01-D-0004
Delivery Order No. 01

Under Contract to:

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30 September 2004

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QUALITY ASSURANCE STATEMENT

Subcontractor: MACTEC Engineering and Consulting, Inc.

Delivery Order Title: Centredale / Woonasquatucket River

Delivery Order No.: 01

Description of audit and review activities:

Presentation of the Interim Final Baseline Ecological Risk Assessment.

QA/QC review activities conducted and documented for all data summary and report preparation activities in accordance with contract requirements.

Description of outstanding issues or deficiencies which may affect data quality:

None identified.

Signature of QA Officer — Date: 27 SEP 04

Subcontractor Project Leader — Date: 8/12/2004

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EXECUTIVE SUMMARY

A Baseline Ecological Risk Assessment (BERA) has been conducted to evaluate current and potential future ecological risk in support of the Remedial Investigation for the Centredale Manor Restoration Project Superfund Site in North Providence, Rhode Island. This BERA has been conducted in accordance with the United States Environmental Protection Agency (USEPA) Ecological Risk Assessment Guidance for Superfund, Process Document for Designing and Conducting Ecological Risk Assessments (USEPA, 1997), as well as USEPA Region I risk assessment guidance contained in Risk Updates (USEPA, 1994, 1995, 1999). USEPA (1997) established an eight-step process for assessing ecological risk. The first two steps are the screening ecological risk evaluation (SERE) with the goals of determining if the site poses no or negligible risk, and identifying which contaminants and exposure pathways require further evaluation. Steps three through seven comprise components of the BERA that is the subject of this report. The SERE conducted in 2000 (TetraTech NUS, Inc., 2000) concluded that the Site had the potential to adversely affect ecological receptors and recommended that a BERA be conducted. As appropriate for a screening level assessment, the SERE compared maximum chemicals of potential concern (COPC) concentrations to conservative screening benchmarks.

The BERA was conducted to provide more realistic estimates of the likelihood and magnitude of potential risk to wildlife receptors associated with the consumption of contaminated prey, drinking water, and incidental ingestion of surface soil and sediment, at the Site; and to other ecological receptors associated with direct contact with and ingestion of surface water, sediment, and floodplain soil. (The specific objectives were to evaluate the six assessment endpoints and associated risk questions described in further detail below). In order to reduce uncertainties associated with the preliminary risk findings from the SERE, a variety of site-specific studies were conducted in preparation for the BERA.

The BERA analyzes potential adverse ecological effects for both current and future conditions caused by hazardous substance releases from a site in the absence of any actions to control or mitigate these releases (i.e., under an assumption of no action at the Site). In accordance with USEPA guidance, the BERA consists of four components: problem formulation, exposure assessment, effects assessment, and risk characterization (including an evaluation of risk uncertainties). Each of these components is described below:

PROBLEM FORMULATION

The problem formulation focuses the BERA and establishes the goals, breadth and major issues for consideration and includes a description of the environmental setting and

resources potentially at risk; the selection of COPCs and their fate, transport, and ecotoxicity; the selection of receptors of concern and species profiles; a conceptual site model (CSM) with complete exposure pathways; assessment and measurement endpoints along with the study rationale, and risk hypotheses.

Environmental Setting

The Woonasquatucket River is designated as a Class B1 waterbody, suitable for primary and secondary human contact recreation and fish and wildlife habitat. The land-use for the eastern shore of Allendale and Lyman Mill reaches is primarily residential with some commercial and industrial activity. Residential, commercial, and industrial properties are located approximately 200 feet or more from the western shore of Allendale and Lyman Mill. Undeveloped land adjacent to the river includes palustrine forest, scrub-shrub, and emergent marsh. Fish and aquatic invertebrates associated with the Woonasquatucket River are typical of a warm-water fishery in New England; these organisms in turn support of variety of wildlife species that specialize on fish, invertebrate, or mixed diets (piscivores, insectivores/vermivores, and omnivores, respectively).

Conceptual Site Model (CSM)

The CSM identifies the source, media, pathway and route of exposures evaluated in the BERA, and the relationship between the measurement endpoints and the assessment endpoints (USEPA, 1997). It serves as a communication tool that illustrates the major pathways by which ecological receptors might be exposed to COPCs associated with releases from the Centredale Manor Site source area. Figure ES-1 presents a generalized CSM for the Site.

Source Area. The main area of the Site, consisting of approximately 9.04 acres, is located in North Providence, Rhode Island, just south of Route 44 on the eastern bank of the Woonasquatucket River (Figure 1). The Site (source area) consists of two parcels located at 2072 and 2074 Smith Street (Lots 200 & 250) that cover approximately 9.04 acres (see Figure 2). The remaining portions of the Site consist of reaches, man-made ponds, and wetlands associated with the Woonasquatucket River. The river flows from north to south (Figure 2).

The Site was used for disposal of wastes containing hazardous substances. Between approximately 1940 and 1970, USEPA has reason to believe that the main area of the Site was the location of a chemical manufacturing facility (which produced hexachlorophene, among other chemicals) and an incineration-based drum recycling facility. Evidence suggests that operations at the former chemical company and drum reconditioning facility resulted in waste disposal onto surface soil and beneath the ground surface. Wastes have also been released directly into the Woonasquatucket River, which runs along the western side of the source area (Tetra Tech NUS Inc., 2000).

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Currently, two high rise buildings (Centredale Manor and Brook Village, both federally-subsidized, senior housing complexes) are located in the main area of the Site, which is zoned for residential occupancy. In addition to the buildings, the main area is covered by roadway, parking lots, and two capped source areas. The first housing complex was built in approximately 1976. The second high rise was built in 1982. Construction records show that hazardous substances were removed from the Site during the construction of the second complex. Samples indicate the presence of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and other contaminants in soil and sediment at the main portion of the Site and the river sediment and floodplain soils downstream. Dioxins and furans have been detected in soils and sediments as well as in fish tissue collected in 1996 from the Woonasquatucket River. Other contaminants detected in Site media include polychlorinated biphenyls (PCBs), volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), hexachloroxanthene (HCX), phthalates, and metals. Elevated VOCs have been detected in overburden groundwater at the Site and are discharging to the adjacent reach of the Woonasquatucket River as confirmed by vapor diffusion samplers (Church et al., 2000).

Migration of Contaminants. Downstream sediments have been impacted through the transport and deposition of contaminants from the source area. Dioxins and furans have been detected in sediments at the Allendale, Lyman Mill and Manton exposure areas (EAs) (Figure 2). With the partial breaching of Allendale Dam in 1991, and the more recent breach in 2001, further contaminant migration may have occurred.

Contaminants that were discharged from the source area entered the Woonasquatucket River and were transported downstream either dissolved in the water column or adsorbed to resuspended sediments. Overland flow during flooding events resulted in contamination of lateral floodplain habitats associated with the river. Over time, contaminants preferentially accumulated in low energy sediment and floodplain soils, which are characterized by higher organic carbon content. Compounds with a propensity to bioaccumulate were taken up by plants, invertebrates, and fish and were transferred through aquatic food webs. Wildlife species that consume these lower trophic level organisms could also be exposed to site-related contaminants. Contaminants that were deposited in floodplain soils could also enter the terrestrial food webs by a similar process.

Potential Ecological Receptors. Potential ecological receptor species considered in this BERA are aquatic and floodplain invertebrates, amphibians, fish, birds, and mammals that depend on aquatic resources of the Woonasquatucket River.

In general, aquatic receptors (including invertebrates and both demersal and pelagic fish species) are exposed to COPCs in sediment and surface water via direct contact, direct ingestion, or by consuming prey items that have bioaccumulated COPCs. Semi-aquatic

receptors (including mammals, birds, reptiles, and amphibians) may be exposed as a result of incidental ingestion of sediment, consumption of water, or ingestion of contaminated prey. Terrestrial invertebrates and wildlife that prey on these species may be exposed to contaminants in floodplain soil directly or by ingesting contaminated prey.

Ecological Exposure Pathways. Ecological receptors may be exposed to site-related contaminants via a variety of exposure pathways. A complete exposure pathway involves a potential for contact between a given receptor and contamination either through direct exposure to an abiotic medium or indirectly through prey consumption. Pathways are evaluated by considering information on contaminant fate and transport, ecosystems potentially affected, and the magnitude and extent of contamination (USEPA, 1997).

The BERA includes evaluation of the following exposure pathways: direct contact with surface water, sediment and floodplain soils by invertebrate receptors; ingestion of biota by piscivorous, insectivorous, and omnivorous wildlife receptors from a background location, a reference area, and the reach of the Woonasquatucket River that constitutes the Site; consumption of drinking water by wildlife receptors; and incidental ingestion of sediment and floodplain soil by wildlife receptors.

Animals and plants that occur in or adjacent to the Woonasquatucket River, including invertebrates, fish, amphibians, birds, and mammals could be exposed to contaminants through contact with floodplain soil, sediment, surface water, and prey consumption. Species representing various trophic levels were selected as representative receptor species to evaluate the assessment endpoints. The selected species are intended to be representative of other species at the same trophic level that share similar ecological characteristics. These groups of species are generally referred to as guilds. By evaluating a representative member of a guild and by accounting for the predominant guilds, the uncertainty associated with missing an important species group or pathway is reduced.

Summary of Data

The data evaluation report indicates the analytical data collected at the Site have undergone data validation procedures consistent with USEPA guidelines (MACTEC, 2003). The data validation activities determined that overall, the data that have been collected meet the data quality objectives (DQOs) for the risk assessment activities. The available data were reviewed to identify those data that were representative of current and potential future site conditions and uses, and that are therefore suitable for evaluating current and potential future ecological risks.

The BERA is based on data collected from several site investigations in soil, surface water, sediment as well as in biota found in and adjacent to the Woonasquatucket River.

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Data were collected from the Centredale Manor Site source area, four reaches of the Woonasquatucket River adjacent to and downstream of the Site (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond), and at an upstream background area (Greystone Mill Pond, which is upstream on the Woonasquatucket River) and a reference area (Assapumpset Brook, which is a tributary that flows into the Woonasquatucket River at Lyman Mill Pond), both of which are unimpacted by the Site).

Dioxins and furans (particularly 2,3,7,8-TCDD), HCX, Aroclor-1254 and possibly PCB-77 appear to be the primary chemical parameters that are detected in environmental media with frequency of detection and concentrations that are indicative of Site-related impacts. In other words, these parameters have clearly elevated concentrations in biota tissue (including fish, earthworms, emerging insects, tree swallow eggs, nestlings, and stomach contents) and sediments in the Site-related exposure areas (Allendale Pond, Lyman Mill Pond, Manton Pond, and Dyerville Pond) compared to the Greystone Mill Pond upstream background area and the Assapumpset Pond and Brook reference area. In addition, a distinctive dioxin/furan "signature" characterized the Site sediment, floodplain soil, white sucker, American eel, largemouth bass, crayfish, emerging insects, earthworms, and tree swallow eggs, nestlings, and stomach contents. In virtually all cases, the average 2,3,7,8-TCDD concentrations are 10 to 100 times higher in Allendale and Lyman Mill compared to the background and reference areas.

Chemicals of Potential Concern (COPC) Selection

Using the data collected in soil, surface water, sediment, and biota (white sucker, largemouth bass, American eel, crayfish, earthworms, emerging insects, and tree swallow eggs, nestlings, and stomach content), chemicals were initially identified as chemicals of potential concern (COPCs) by media for the site and chemicals of interest (COIs) in reference/background areas. COPCs require further evaluation in the risk assessment if the chemical concentrations are above risk-based screening concentrations.

COPCs selected for the environmental media include the following:

1. Surface water: bis-2-ethylhexylphthalate, several pesticides, inorganics (total and dissolved), ammonia, dioxins and furans, and HCX.
2. Overburden groundwater: COPCs include two VOCs and six semivolatile organic compounds (SVOCs).
3. Sediment: SVOCs, particularly PAHs, pesticides, inorganics, PCBs, and dioxins, furans, and HCX.
4. Crayfish tissue: Pesticides, inorganics, PCBs, and dioxins, furans, and HCX.
5. Emerging insects: PCBs, and dioxins, furans, and HCX.
6. Fish tissue: SVOCs (particularly PAHs), pesticides, inorganics, PCBs, and dioxins, furans, and HCX.

7. Floodplain soil: SVOCs (particularly PAHs), pesticides, inorganics, PCBs, and dioxins, furans, and HCX.
8. Earthworm tissue: Pesticides, inorganics, PCBs, and dioxins, furans, and HCX.
9. Tree swallow egg tissue: Pesticides, PCBs, and dioxin, furans, and HCX.
10. Tree swallow nestling tissue: Pesticides, PCBs, and dioxin, furans, and HCX.

The list of COIs for the background and reference areas is very similar to that for the Site. For the background area and reference area sediment COPCs include SVOCs, particularly PAHs, pesticides, inorganics, PCBs, and dioxins, furans, and HCX. In fish tissue, COPCs also include SVOCs, particularly PAHs, pesticides, inorganics, PCBs, and dioxins and furans. HCX was not detected and therefore not selected as a COI for white sucker, but HCX was selected as a COPC for American eel and largemouth bass. In surface water, COPCs include no VOCs, no SVOCs, no pesticides, inorganics (total and dissolved), and dioxins and furans. Due to differences in the selection criteria and environmental media evaluated, the COPCs selected in the BERA are not identical to those retained in the Baseline Human Health Risk Assessment (BHHRA)

Ecotoxicology of Selected COPCs

The BERA summarized the available toxicological literature for all classes of compounds identified as COPCs for each evaluated receptor category (i.e., invertebrates, fish, birds, and mammals). The Toxic Equivalency (TEQ) approach was employed in the BERA to overcome the difficulty in assessing the overall toxicity of dioxin mixtures. Specific Toxicity Equivalency Factors (TEFs) have been developed for human/mammals, birds, and fish. A Toxic Equivalent TEQ concentration was derived by summing the products of the individual congener concentrations and their corresponding TEFs for a given environmental sample. The TEQ values presented in the BERA are reported separately for the sum of dioxin and furan congeners and the sum of PCB congeners.

No TEF has been established for HCX and interim values for mammals, birds, and fish were developed as part of the BERA based on a biochemical competition assay to assess the relative affinity of HCX relative to TCDD for binding to both fish (trout) and mammal (human) Ah-receptors (Hahn, 2001). The selected interim TEF (0.0002) for HCX was supported by both *in vitro* and *in vivo* laboratory studies reported in the literature.

Assessment and Measurement Endpoints

The assessment endpoints evaluated in the BERA consist of the following:

1. Protection and maintenance (i.e., survival, growth, reproduction) of aquatic and floodplain invertebrate communities which are a forage base for fish and wildlife.

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2. Protection and maintenance of demersal, omnivorous fish populations as a forage base or sport fishery.
3. Protection and maintenance of pelagic, piscivorous or semi-piscivorous fish populations as a forage base or sport fishery.
4. Protection and maintenance of piscivorous mammal and bird populations.
5. Protection and maintenance of insectivorous mammal and bird populations.
6. Protection and maintenance of omnivorous mammal and bird populations.

Assessment Endpoint 1 was segregated into two separate components (i.e., 1a and 1b; focused on the aquatic and floodplain invertebrates communities, respectively) to facilitate the risk analysis. For Assessment Endpoint 5, the bird and mammal species that forage on flying insects (true insectivores) were distinguished from those that feed primarily on earthworms (i.e., vermivores).

Risk questions and measurement endpoints were identified for each of these six assessment endpoints:

Assessment Endpoint 1a. Protection and maintenance (i.e., survival, growth, reproduction) of aquatic invertebrate communities, which are a forage base for fish and wildlife.

The following risk questions were established for this assessment endpoint:

A. Do measured concentrations of COPCs in surface water exceed appropriate criteria and/or guidelines for the protection of aquatic life, with special consideration of reproduction and early lifestage survival?
B. Do measured concentrations of COPCs in whole sediment exceed appropriate guidelines for the protection of benthic macroinvertebrate populations?
C. Are sediments toxic to aquatic macroinvertebrates under chronic laboratory exposure conditions?
D. Do measured and modeled concentrations of COPCs in the tissues of aquatic macroinvertebrates (such as aquatic insects, crayfish, and molluscs) exceed benchmarks for residue effects on survival, growth or reproduction?
E. Do the available aquatic macroinvertebrate data indicate presence/absence of ecological integrity due to VOC discharge in riffle/run habitat adjacent to the Site?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of surface water COPC concentrations to criteria/guidelines.
B. Comparison of sediment COPC concentrations to benchmarks/guidelines.
C. Site-specific whole sediment laboratory bioassays.

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D. Comparison of measured COPC concentrations in aquatic macroinvertebrates to Critical Body Residues (CBRs).
E_1. Site-specific study of aquatic macroinvertebrate community structure/function.
E_2. Site-specific study of emerging aquatic macroinvertebrate productivity.

Assessment Endpoint 1b. Protection and maintenance (i.e., survival, growth, reproduction) of floodplain invertebrate communities, which are a forage base for wildlife.

The following risk questions were established for this assessment endpoint:

A. Do measured concentrations of COPCs in floodplain soil exceed appropriate guidelines for the protection of floodplain soil invertebrate populations?
B. Do measured and modeled concentrations of COPCs in the tissues of floodplain soil invertebrates (such as earthworms) exceed benchmarks for residue effects on survival, growth or reproduction?
C. Do the available floodplain soil invertebrate data indicate presence/absence of ecological integrity?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of floodplain soil COPC concentrations to benchmarks/guidelines.
B. Comparison of measured site-specific COPC concentrations in floodplain soil invertebrates to CBRs.
C. Site-specific study of floodplain soil invertebrate community structure/function.

Assessment Endpoint 2. Protection and maintenance of demersal, omnivorous fish populations as a forage base or sport fishery.

The following risk questions were established for this assessment endpoint:

A. Do field surveys of demersal fish populations indicate presence/absence of ecological integrity?
B. Do measured and modeled concentrations of COPCs or TEQs in eggs and/or tissues of demersal, omnivorous fish (such as carps, suckers, other cyprinids, bullheads, turtles or American eel) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

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A_1. Fish length-weight relationships and condition indices relative to habitat characterization.
A_2. Percent gross lesions in individual fish.
A_3. Demographic structure analysis of dominant species.
A_4. Species richness and relative abundance of ichthyoplankton.
B_1. Comparison of measured concentrations or toxic equivalencies in fish tissue to literature derived CBRs.
B_2. Comparison of modeled concentrations or toxic equivalencies in eggs and fish tissue to site-specific CBRs.
B_3. Partial life cycle laboratory bioassay using channel catfish eggs, embryos, and fry.

Assessment Endpoint 3. Protection and maintenance of pelagic, piscivorous or semi-piscivorous fish populations as a forage base or sport fishery.

The following risk questions were established for this assessment endpoint:

A. Do field surveys of piscivorous fish populations indicate presence/absence of ecological integrity?
B. Do measured and modeled concentrations of COPCs or TEQs in eggs and/or tissues of pelagic, piscivorous or semi-piscivorous fish (such as sunfishes, other centrarchids, or yellow perch) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A_1. Fish length-weight relationships and condition indices relative to habitat characterization.
A_2. Percent gross lesions in individual fish.
A_3. Demographic structure analysis of dominant species.
A_4. Species richness and relative abundance of ichthyoplankton.
B_1. Comparison of measured concentrations or toxic equivalencies in fish tissue to literature derived CBRs.
B_2. Comparison of modeled concentrations or toxic equivalencies in eggs and fish tissue to site-specific CBRs.

Assessment Endpoint 4. Protection and maintenance of piscivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

A. Do ingestion doses of COPCs in piscivorous wildlife (such as mink, river otter, kingfisher, great blue heron, or American bald eagle) exceed Toxic Reference Values (TRVs) or TEQs for adverse effects on survival, growth, or reproduction?
B. Do residues of COPCs in tissues of piscivorous wildlife (such as mink, river otter, kingfisher, great blue heron, or American bald eagle) exceed benchmarks for adverse effects on survival, growth, or reproduction?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of estimated ingestion doses in piscivorous wildlife with TRVs and toxic equivalencies.
B. Comparison of estimated piscivorous wildlife residues with CBRs.

Assessment Endpoint 5. Protection and maintenance of insectivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

A. Do ingestion doses of COPCs in insectivorous wildlife (such as shrew, bat, American woodcock or tree swallow) exceed TRVs or TEQs for adverse effects on survival, growth or reproduction?
B. Do residues of COPCs in eggs and/or tissues of insectivorous wildlife (such as shrew, little brown bat, American woodcock, or tree swallow) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?
C. Do field survey data on insectivorous wildlife population indicate presence/absence of ecological integrity?
D. Do mix-function oxidase (MFO) liver enzyme levels in swallow nestlings indicate that they have been exposed to compounds with "dioxin"-like metabolic activity?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of estimated ingestion doses in insectivorous wildlife with TRVs and toxic equivalencies.
B_1. Comparison of measured insectivorous wildlife tissue and/or egg residues with CBR data.
B_2 Comparison of estimated insectivorous wildlife tissue and/or egg residues with site-specific CBR data.
C_1 Site-specific measurement of reproductive effects in local tree swallow populations.
C_2 Site-specific survey of calling amphibians.
D. Elevated MFO activity in tree swallow nestling liver tissue.

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Assessment Endpoint 6. Protection and maintenance of omnivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

A. Do ingestion doses of COPCs in omnivorous wildlife (such as raccoon or mallard) exceed TRVs or TEQs for adverse effects on survival, growth or reproduction?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of estimated ingestion doses in omnivorous wildlife with TRVs and TEQs.

EXPOSURE ASSESSMENT

The objective of the exposure assessment is to estimate the type and magnitude of an ecological receptors' exposures to COPCs at or migrating from the site. The exposure assessment is conducted to: 1) characterize the relevant exposure areas, exposure pathways, and receptors and 2) identify the concentration, or dose, of COPCs that receptors may receive through the identified exposure pathways.

Identification of Exposure Areas

Based on the presence of current or historical impoundments in the Woonasquatucket River, the following exposure areas were identified:

- EA1 is referred to as the Allendale EA;
- EA2 is referred to as the Lyman Mill EA;
- EA3 is referred to as the Manton EA;
- EA4 is referred to as the Dyerville EA;
- The upstream background area is referred to as the Greystone background; and,
- The reference area is referred to as the Assapumpset Brook and Pond reference area.

Exposure Point Concentrations

For each contaminant selected as a COPC, at each exposure point, representative concentrations in fish (American eel, largemouth bass, and white sucker), submerged sediment, surface water, and bank soil were identified as the basis for the exposure assessments. The representative concentrations (exposure point concentrations or EPCs)

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were calculated based on the 95% upper confidence limit (UCL) on the arithmetic mean concentration of the data. The procedures used to identify the 95% UCL and the EPC were selected based on the size of the data set and the distribution type for the concentration data. Reasonable Maximum Exposures (RME) expressed as the highest estimate of exposure were evaluated in the risk assessment. In addition, exposures were assessed using the Central Tendency (CT), which represents the typical or average exposure conditions; however, the exposures are not actually measured and there is variability among receptors which might be present at the Site with respect to frequency and duration of exposure and their consumption rates.

Identification of Exposure Models and Parameters

Chemical-specific intakes were calculated in a manner consistent with USEPA guidance for risk assessment. Estimated Daily Intakes (EDIs) of COPCs were calculated as the measure of exposure for each selected wildlife receptor. The EDIs are expressed as milligrams of contaminant per kilogram of bodyweight per day (mg/kg/day). The following exposure parameters are included in the dose calculations employed in the BERA:

- Concentrations in biota tissue, sediment, surface water, and floodplain soil (C)
- Consumption rate (IR)
- Exposure frequency (EF)
- Fraction ingested from contaminated source (FI)
- Exposure duration (ED)
- Body weight (BW)

EFFECTS ASSESSMENT

The purpose of the effects assessment is to characterize the relationship between the concentration or dose of COPC administered or received and the incidence of adverse effects in the ecological endpoint receptor at the appropriate level of ecological organization (i.e., usually population- or community-level). The following types of information were used in the BERA:

- Toxicity Reference Values (TRVs) that relate a threshold concentration or ingested dose to an adverse and relevant biological response. TRVs were established for surface water, sediment, floodplain soil, and biological tissue (including invertebrates, fish, birds, and mammals)
- Sediment bioassay
- Aquatic macroinvertebrate community study
- Study of emerging insect productivity
- Floodplain macroinvertebrate community study

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- Fish community survey (including fish length-weight relationships, percent gross lesions in individual fish, demographic structure analysis, and fish species richness)
- Fish ichthyoplankton study
- Early Lifestage (ELS) laboratory bioassay
- Multiple year tree swallow reproductive and nestling study
- Multiple year amphibian call survey

It is important to note that all but the first type are Site-specific measures of effect derived specifically for this BERA.

MAJOR BERA UNCERTAINTIES

Table ES-1 summarizes the principal risk uncertainties and identifies the relevant assessment endpoint(s) and COPCs, as well as the direction (i.e., under- or over-estimating risks) and magnitude of likely effect. Those uncertainties that likely resulted in the BERA risk estimates being substantially under-estimated are indicated by shading: these include: (i) not deriving TCDD plant tissue concentrations to evaluate omnivorous mammal risks, (ii) not characterizing floodplain exposures in the Manton and Dyerville EAs, (iii) the lack of critical body residue data for certain COPCs, and (iv) the limited ichthyological survey results. On the other hand, the following categories probably resulted in risk estimates generally being over-estimated in the BERA: (i) estimated fish tissue EPCs for Dyerville, (ii) the use of relatively large fish in the tissue sampling program, and (iii) the use of standard benchmarks as one line of evidence for several of the assessment endpoints.

Due to the magnitude of risk estimates and the concurrence of multiple lines of evidence, which included extensive site-specific biological effects studies, it is concluded that the identified uncertainties are unlikely to affect the overall conclusions derived in the BERA.

MAJOR FINDINGS

Table ES-2 summarizes the results of the BERA for each of the assessment endpoints. The major findings of the BERA include the following:

- Although HCX was assumed to be the major risk driver in the SERE (TEF assumed to be 1), it was an insignificant contributor to the risk findings in the BERA based on use of a more realistic interim TEF value derived as part of this study and as supported by the findings of ELS bioassay.
- The benthic macroinvertebrate community that occurs in low-gradient habitats (lentic) upstream of impoundments within the study area is at substantial risk of

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harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.

- However, the discharge of VOC-contaminated groundwater into the reach of the Woonasquatucket River adjacent to the Site does not appear to have adversely affected the benthic macroinvertebrate community associated with lotic (riffle/run) habitats.
- The soil invertebrate community occurring within the Woonasquatucket River floodplain at the study area is not likely at substantial risk of harm due to exposure to Site-related contaminants in floodplain soil or tissue residues.
- Demersal, omnivorous fish populations that occur within the study area may be at substantial risk of harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.
- The time course of ELS mortality (i.e., majority within a week of exposure) in the laboratory bioassay may corroborate findings of the ichthyoplankton survey; limited evidence of facial-cranial malformations was observed in both the laboratory and field studies.
- Pelagic, piscivorous or semi-piscivorous fish populations that occur within the study area may be at substantial risk of harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.
- Piscivorous mammal and bird populations that occur within the study area may be at substantial risk of harm due to direct exposure to Site-related contaminants in surface water, sediment, and prey. In addition, consumption of contaminated fish may result in elevated tissue residues in these receptors resulting in adverse reproductive effects (i.e., bioaccumulation hazard).
- Dioxins and furans (particularly 2,3,7,8-TCDD), coplanar PCBs, Aroclor-1254, 4,4'-DDE, and methylmercury are the largest contributors to the estimated risk for piscivorous wildlife species.
- Insectivorous and vermivorous mammal and bird populations that occur within the study area appear to be at substantial risk of harm due to direct exposure to Site-related contaminants in surface water, sediment, floodplain soil, and prey. In addition, consumption of contaminated insect and earthworm prey may result in elevated tissue residues in these receptors resulting in adverse reproductive effects (i.e., bioaccumulation hazard).
- The large and positive accumulation rates of TCDD in swallow nestlings and induction of elevated EROD activity in their livers are strongly indicative of local contamination from the individual EAs where the birds nested because of the limited foraging behaviors of the adult birds. Moreover, the swallow study also demonstrated a strong correlation between TCDD concentrations in tree swallow nestling tissue and stomach contents.
- Dioxins and furans (particularly 2,3,7,8-TCDD) and coplanar PCBs are the primary contributors to risks for insectivorous wildlife that consume insects emerging from the Woonasquatucket River.

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- A strong and consistent negative relationship between the TCDD (and TEQ) concentration in swallow eggs and percent hatchability was demonstrated.
- Dietary exposures and measured or modeled tissue burdens in avian eggs and mammal whole body tissue pose a substantial risk of harm to piscivorous and insectivorous wildlife species.
- Omnivorous mammal and bird populations that forage within the study area are not at substantial risk of harm due to exposure to Site-related contaminants in surface water, floodplain soil, and terrestrial prey items. However, omnivorous mammals could be adversely affected as a result of exposure to Site-related contaminants in sediment and aquatic prey. Although the exclusive use of aquatic habitat by omnivorous mammals, such as the raccoon, could result in substantial population-level effects, the spatially and temporally varied diets and exposures of these receptors minimizes the likelihood that demographically significant effects would occur.

The concentrations in biota tissue of the predominant risk contributors are directly related to corresponding sediment concentrations. Although direct contact exposures to sediments are not associated with the largest risks at the Site (i.e., COPCs pose primarily a bioaccumulation hazard and risks are associated with the indirect trophic transfer of COPCs), the sediments appear to be associated with the largest risks at the Site.

NEXT STEPS

The Remedial Investigation (RI) and Feasibility Study (FS) will soon be completed. The RI will: determine and summarize the sources, nature and extent of contamination at the Site; characterize the fate and transport of contaminants; and evaluate potential human health and ecological risks resulting from exposure to Site-related contaminants. The FS will evaluate risk management strategies and alternatives for remediating contamination that is found to pose an unacceptable risk to human health or the environment. The FS will also evaluate the long-term effectiveness of the short-term removal actions and determine whether additional action is required to affect a permanent remedy.

In support of the FS, Preliminary Remediation Goals (PRGs) will be estimated for Chemicals of Concern or COCs (those chemicals that are associated with unacceptable risk to ecological receptors) by a two-step process. In the first step, biota tissue concentrations for the most significant COPCs will be identified for various HQs of 0.1, 1, and 10. In the second step, using the concept of steady-state conditions between lipophilic substances in sediment and fish tissue, sediment concentrations corresponding to the risk-based tissue concentrations will be identified.

Development of the ecological PRGs will be discussed in further detail in a separate document. The calculated risks for the reference area and background area will

obviously be a consideration in the derivation of PRGs and the selection of remedial objectives.

The results of the RI and FS will be used to formulate a Proposed Plan for the Site. The Proposed Plan will recommend remedial actions that will result in overall protection of human health and the environment, fulfill Superfund requirements, be acceptable to stakeholders, and satisfy USEPA remedial guidelines. USEPA's remedial action at this site will allow the Woonasquatucket River and associated reaches and impoundments to return to a fishable and swimmable condition.

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1.0 INTRODUCTION

The U.S. Environmental Protection Agency (USEPA) Region I and U.S. Army Corps of Engineers (USACE) are conducting a Baseline Ecological Risk Assessment (BERA) and a Baseline Human Health Risk Assessment (BHHRA) for the Centredale Manor Restoration Project Superfund Site located in North Providence, Rhode Island (hereafter referred to as "the Site"). The site locus is indicated in Figure 1, and Figure 2 provides a schematic layout of the entire study area.

This BERA has been developed to reflect the goals and objectives discussed and agreed upon during meetings with USEPA, USACE, Battelle, and Harding ESE (now MACTEC Engineering and Consulting, Inc.) conducted on July 17, 2000, November 17, 2000, December 4, 2000, December 19, 2000, and February 28, 2000, and presented in meeting minutes and other support documents. This report was prepared to be consistent with the requirements set forth in the Statement of Work (SOW) and the Addendum to the SOW prepared by USEPA (USEPA, 2000a and USEPA, 2000b). The BERA is based on data obtained during the Remedial Investigation (RI) site characterization and sampling activities. The BERA, BHHRA, and RI summarize existing conditions at the Site with respect to exposure to and effects of contaminants in sediment, surface water, groundwater, surface soil, and biota. These documents provide the data and analysis necessary to support risk management decisions for the Site. USEPA's remedial action at this site will allow the Woonasquatucket River and associated reaches and impoundments to return to a fishable and swimmable condition.

Contaminants that are present in surface water and aquatic sediment may have bioaccumulated in fish and other biota present in the Woonasquatucket River. Aquatic receptors (including invertebrates and both demersal and pelagic fish species) are exposed to Contaminants of Potential Concern (COPCs) in sediment and surface water via direct contact, direct ingestion, or by consuming prey items that have bioaccumulated contaminants. Semi-aquatic receptors (including mammals, birds, reptiles, and amphibians) may be exposed as a result of incidental ingestion of sediment, consumption of water, or ingestion of contaminated prey. Terrestrial invertebrates and wildlife that prey on these species may be exposed to contaminants in floodplain soil directly or by ingesting contaminated prey.

The approach used in conducting the BERA followed an established framework and guidelines for assessing ecological risks. Specifically, the USEPA guidance for ecological risk assessment, *Ecological Risk Assessment Guidance for Superfund, Process Document for Designing and Conducting Ecological Risk Assessments* (1997), established an eight-step process for assessing ecological risk. The first two steps are the screening level evaluation with the goals of determining if the site poses no on negligible risk, and identifying which contaminants and exposure pathways require further

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evaluation. The results of the screening level evaluation (TetraTech NUS, Inc., 2000) are discussed below. Steps 3 through 7 comprise components of the BERA. Step 3 involved the refinement of the SERE Problem Formulation, resulting in the identification of six assessment endpoints and a number of specific risk questions for each. Steps 4 and 5 involved identifying the types, quality, and implementability of ecological data necessary to evaluate the assessment endpoints identified in Step 3; these steps resulted in the preparation of a Project Work Plan (Harding ESE, 2001a) and a Field Sampling Plan (Harding ESE, 2001b). Step 6 consisted of the Site investigation activities that were conducted during the summer of 2001 and subsequent data analysis activities, and the preparation of this document completes Step 7. Step 8 will consist of risk management activities undertaken for the Site.

The Streamlined Ecological Risk Evaluation (SERE) was conducted in 2000 to evaluate the distribution of COPCs in the Woonasquatucket River and adjacent floodplain and river bank soils (Tetra Tech NUS, Inc., 2000). The SERE comprises Steps 1 and 2 of the eight step process for evaluating ecological risk under the Superfund Program (USEPA, 1997). The primary goals of the SERE were to determine whether any portion of the Site represented an obvious threat to ecological health (i.e., "hot spots"), where a Non-Time Critical Removal Action (NTCRA) might be warranted, and to identify those site contaminants that do not pose a threat to the ecosystem as determined using conservative screening assumptions. The reach of the Woonasquatucket River evaluated was between the Route 44 Bridge and the Lyman Mill Dam.

The SERE evaluated available sediment, surface water, and surface soil samples and compared summarized analytical chemistry data to screening ecological benchmarks developed for aquatic life, terrestrial invertebrates, terrestrial plants, and wildlife. Analytical data were segregated into a number of sample groupings including residential soil, bank soil, floodplain soil, floodplain sediment, aquatic sediment, source area surface water, floodplain surface water, and downstream surface water. Conservative screening benchmarks were developed for the following receptor groups: soil invertebrates, terrestrial vegetation, herbivorous birds, and mammals, carnivorous birds and mammals, omnivorous birds and mammals, benthic invertebrates, fish, amphibians, and reptiles.

The SERE evaluated the following four Assessment Endpoints:

- Aquatic and floodplain invertebrate communities exposed to surface water and sediment;
- Soil invertebrate and plant communities exposed to surface soil;
- Piscivorous mammal and bird populations exposed to sediment; and,
- Insectivorous and herbivorous mammal and bird populations exposed to surface soil.

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The SERE identified the widespread presence of contaminants at concentrations above screening benchmarks and concluded that the majority of contaminants detected in environmental media evaluated could not be eliminated from further consideration. COPCs were identified for soil, sediment, and surface water media and included dioxin congeners, 1,2,4,5,7,8-hexachloroxanthene (HCX), SVOCs, pesticides, PCB mixtures, and various inorganic analytes. No contaminant hot spots were identified in the SERE but the majority of detected SVOCs, PCBs/pesticides, and inorganic analytes were retained for further evaluation in the BERA (Tetra Tech NUS, 2000). The SERE also concluded that upstream conditions (principally the Smithfield WWTP, could contribute substantially to ecological risks at the Site (Tetra Tech NUS, 2000). The SERE assumed a Toxicity Equivalency (TEF) of 1 for HCX (i.e., assumed that HCX was as toxic as 2,3,7,8-TCDD).

The BERA was conducted to provide more realistic estimates of the likelihood and magnitude of potential risk to wildlife receptors associated with the consumption of contaminated prey, drinking water, and incidental ingestion of surface soil and sediment, at the Site; and to other ecological receptors associated with direct contact with and ingestion of surface water, sediment, and floodplain soil. (The specific objectives were to evaluate the six assessment endpoints and associated risk questions identified in the project Scope of Work [SOW] and described in further detail below). In order to reduce uncertainties associated with the preliminary risk findings from the SERE, a variety of site-specific studies were conducted in preparation for the BERA.

The following Site-specific studies were conducted to support the BERA and to reduce uncertainties in the risk estimates:

- AhR biochemical binding assay – (to estimate a more realistic interim TEF for HCX)
- Chronic sediment bioassay – (to reduce benchmark uncertainties)
- Aquatic macroinvertebrate community study – (to reduce benchmark uncertainties)
- Emerging insect productivity study – (to reduce benchmark uncertainties)
- Floodplain macroinvertebrate community study – (to reduce benchmark uncertainties)
- Fish community survey – (including fish length-weight relationships, percent gross lesions in individual fish, demographic structure analysis, and fish species richness) – (to provide population- and community-level data on ecological integrity to a potentially sensitive group of receptors)
- Fish ichthyoplankton study – (to assess potential recruitment effects for sensitive life stages)
- Early Lifestage (ELS) laboratory bioassay – (to evaluate potential toxicity associated with HCX exposure and provide corroboration of ichthyoplankton study)

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- Tree swallow reproductive and nestling study – (multiple year population-level study for receptor with sensitive life stage and substantial exposures)
- Multiple year amphibian call survey – (assess potential toxicity to a receptor group with limited toxicological data)

This BERA is divided into 11 sections: Section 1 consists of this introduction; the problem formulation is presented in Section 2, and Sections 3 through 9 present the risk evaluation for each of the six assessment endpoints (i.e., specific BERA objectives) identified in the Project Work Plan. Section 10 discusses general BERA uncertainties and risk assessment conclusions are summarized in Section 11.

Appendix A addresses sample selection considerations for the risk assessment. Appendix B contains the photographs of site features to support the exposure assessment. Responses from trustee agencies concerning the presence of threatened or endangered species are provided in Appendix C. Appendix D provides summaries of ecological benchmarks and reference doses, and various bioassessment reports and a sediment toxicity test report are presented in Appendix E and F, respectively. Biota tissue residue effects data are summarized in Appendix G; Appendix H includes an early life stage (ELS) toxicity report, and receptor-specific exposure parameters and chemical specific bioaccumulation factors are presented in Appendix I and J, respectively. Appendices K, L, and N present the food chain exposure modeling results for piscivores, insectivores, and omnivores, respectively, and Appendix M provides the two tree swallow population study reports.

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2.0 PROBLEM FORMULATION

The problem formulation focuses the BERA and establishes the goals, breadth and major issues for consideration (USEPA, 1997, 1998). It contains sections on the environmental setting and resources potentially at risk; the selection of COPCs and their fate, transport, and ecotoxicity; the selection of receptors of concern and species profiles; a conceptual site model (CSM) with complete exposure pathways; assessment and measurement endpoints along with the study rationale, and risk hypotheses. The CSM for the BERA is presented in Figure 3.

2.1 ENVIRONMENTAL SETTING

The Woonasquatucket River flows 18 miles from North Smithfield to the Upper Narragansett Bay draining an area of approximately 51.9 square miles. At its confluence, the river joins with the Moshassuck River in downtown Providence forming the Providence River. It is currently the focus of urban revitalization efforts and was recognized as an American Heritage River in August 1998. The upper section of the river is relatively pristine and rural; however, the urban portion of the river below the Smithfield line has been impacted by sewage and industrial waste for more than a century. The reach of the river that includes the Site is classified as an "impaired water" under Section 303(d) with the cause of impairment attributed to pathogens, cadmium, copper, lead, PCBs, mercury, dioxins, depressed dissolved oxygen, and elevated nutrient levels (RIDEM, 2001). The Smithfield Wastewater Treatment Plant (WWTP) (which is located immediately approximately 0.5 miles upstream of Greystone Mill Pond) has been issued a Rhode Island Pollutant Discharge Elimination System (RIPDES) discharge permit (as a component of a TMDL implementation for the reach) to control the release of nutrients contributing to the hypoxia and excessive algal growth characteristic of summer conditions in the river. Historically, discharges from the WWTP have resulted in a two- to three-fold increase in the concentrations of total nitrogen (as nitrate) and phosphorus down to Allendale Dam; in this reach, phosphorus concentrations are elevated enough to account for the evident eutrophication characteristic of Greystone, Allendale and Lyman Mill Ponds (Berger, 2000). Another unidentified nutrient input has also been documented for the river reach below Lyman Mill. Discharge from the WWTP also contributes loadings of copper, lead, mercury, nickel, silver, and zinc to the river (Berger, 2000).

The site includes the reach of the Woonasquatucket River from the Route 44 Bridge, immediately upstream of the Centredale Manor apartment complex, downstream to the former Dyerville Dam. This comprises a linear reach of the river of approximately 3 miles (Figure 2). The area surrounding the Site is comprised of residential, commercial, and light industry properties. Generally, residential and commercial properties abut the eastern boundary of the Site and the Woonasquatucket River (North Providence, Rhode

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Island), while industrial and commercial properties abut the western river boundary in Johnston, Rhode Island. Figures 4 and 5 depict habitat types (Cowardin et al., 1979) and land use in and surrounding the Site. Land use information for the surrounding area was incorporated into the figures from the Rhode Island RIGIS database and available digitized aerial photographs.

As discussed in the Work Plan and FSP (Harding ESE, 2001a,b), the study area was segregated into general EAs as defined by the four historical impoundments in the Woonasquatucket River (see Figure 2). EA1 consists of the Centredale Reach portion of Woonasquatucket River, Allendale Pond floodplains, and the Allendale Pond channel; hereinafter referred to as "Allendale". The Lyman Mill reach of the Woonasquatucket River and the Lyman Mill Pond represent EA2; hereinafter referred to as "Lyman Mill". EA3 consists of the reach of the Woonasquatucket River between Lyman Mill Dam and the Manton Mill Pond Dam (hereinafter referred to as "Manton") and the Dyerville reach of the Woonasquatucket River and Dyerville Pond constitute EA4 (hereinafter referred to as "Dyerville"). Background samples were collected from the Woonasquatucket River upstream of the Site (including Greystone Mill Pond), and reference samples were collected from Assapumpset Pond and Brook; these areas are hereinafter referred to as "Greystone" and "Assapumpset", respectively.

EA1 consists of the Centredale Reach portion of the Woonasquatucket River, the Allendale Pond floodplains, and the Allendale Pond channel. The Lyman Mill reach of the Woonasquatucket River and the Lyman Mill Pond represent EA2. Manton Pond is EA3 and the Dyerville reach of the Woonasquatucket River and Dyerville Pond constitute EA4. Background/reference sample collection areas include the Woonasquatucket River upstream of the Site (including Greystone Mill Pond), and Assapumpset Pond and Brook. In this BERA, the term "exposure area" has been used to identify separate areas of exposure which area based on the presence of current or historical impoundments to the Woonasquatucket River. The exposure areas correspond to the EAs identified above as follows:

- EA1 is referred to as the Allendale (or APB) EA
- EA2 is referred to as the Lyman Mill (or LPX) EA
- EA3 is referred to as the Manton (or MAP) EA
- EA4 is referred to as the Dyerville (or DYP) EA
- The upstream background area is referred to as the Greystone (or GMP) background area
- The reference area is referred to as the Assapumpset (or RAB) reference area.

The ecological habitat types associated with the Site are characteristic of fragmented, disturbed, and developed landscapes in the New England region and include riverine, lacustrine and palustrine systems (*sensu* Cowardin). The following briefly describes the

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ecological habitats associated with each exposure area (including the reference area). Photographs of each of these areas are provided in Appendix B.

Greystone. This section of the study area is the upstream background area. This area includes a short section of the river above Greystone Mill Pond to the Smithfield WWTP, Greystone Mill Pond, and river reach below the dam to Route 44. Riverine and palustrine wetland types dominate this section of the river. The riverine sections include the river above Greystone Mill Pond and the section of river below the Greystone Mill Pond dam. The river above Greystone Mill pond is consistent with a lower perennial riverine system (i.e., slow flow and gradient), with well developed floodplains, bordered by palustrine scrub shrub and emergent wetlands. The river below the dam is relatively shallow with sections of higher flow and gradient (i.e., riffle and run). The bottom substrate varies from rock cobble bottom to unconsolidated sand and muck. Water depths range from several inches to approximately six feet. The river is bordered by palustrine forest and emergent wetlands are present along western edge of the river, while the eastern edge is dominated by palustrine wetlands. Representative photographs of the Greystone upriver background area are provided in Appendix B.

The Greystone Mill Pond is characterized as palustrine unconsolidated bottom and aquatic bed wetland. Dense rooted submergent vegetation (i.e., water milfoil [*Myriophyllum* sp.]) dominates the bottom of the pond; however, there are large areas of unvegetated muck, mostly in front of the dam. Aquatic vegetation above the dam includes rooted submergent, emergent and floating leafed plants including water milfoil, water lilies (*Nuphar* and *Nymphaea* spp.), and duck weed (*Lemna* sp.). The palustrine scrub shrub wetlands bordering the upper reaches of the river and Pond are dominated by water willow (*Decodon verticlatus*), alders (*Alnus* sp.), sweet pepper bush (*Clethra alnifolia*), willows (*Salix* spp.), birches (*Betula* sp.) and red maples (*Acer rubrum*). Avian wildlife observed in the upper sections of the river and Pond included tree swallows (*Tachycineta bicolor*), mallard (*Anas platyrhynchos*), black duck (*A. rubripes*), wood duck (*Aix sponsa*), mute swan (*Cygnus olor*), Canada geese (*Branta canadensis*), belted kingfisher (*Ceryle alcyon*), green heron (*Butroidies striatus*), and great blue heron (*Ardea herodias*). Other wildlife observed using these areas included muskrat (*Ondatra zibethica*) and raccoon (*Procyon lotor*). Reptiles and amphibians observed in the lacustrine and riverine sections included painted and snapping turtles (*Chrysemys picta* and *Chelydra serpentina*, respectively) and frogs (*Rana* spp.). Fish species observed include chain pickerel (*Esox niger*), brown bullhead (*Ameiurus nebulosus*), yellow bullhead (*A. natalis),* white sucker (*Catostomus commersoni*), tessellated darter (*Etheostoma olmstedi*), and American eel (*Anguilla rostrata*).

Allendale. This section encompasses the river section between the Route 44 bridge and Allendale Dam. The upper portion of this reach is characterized as upper perennial riverine habitat dominated by rock bottom. There is very little emergent or submergent vegetation associated with this section of the river. The western bank of the river is

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relatively steep and undercut and dominated by trees and scrub shrub vegetation. The eastern bank of the river is mostly developed (i.e., Brookside Village and Centredale Manor) and included rip rap along two capped areas. Several muskrats and mallard ducks were observed in this section of the river. Representative photographs of the Allendale EA are provided in Appendix B.

The lower portion of the reach is characterized as riverine wetland habitat dominated by unconsolidated bottom (e.g., sand and muck) bordered by palustrine emergent and scrub shrub wetland habitat. Vegetation includes yellow birch (*Betula alleghaniensis*), gray birch (*Betula populifolia*), dogwood (*Cornus amomum*), black willow (*Salix nigra*) and red maple. The palustrine emergent wetland is dominated by jewel weed (*Impatiens capensis*), smart weed (*Polygonum* spp.), nettle (*Laportea* sp.), cattails (*Typha* sp.), wool grass (*Scirpus cyperinus*), and purple loosestrife (*Lythrum salicaria*). Wildlife observed using these areas included tree swallows, killdeer (*Charadrius vociferus*), Canada geese, mourning doves (*Zenaida macroura*), muskrat, and mink (*Mustela vison*). Reptiles and amphibians including snapping turtles, and green frogs (*Rana clamitans*) were also observed utilizing this area. Fish observed in this reach include American eel, brown bullhead, tessellated darter, and white sucker.

Lyman Mill. This river reach is bounded by the Allendale dam upstream and the Lyman Mill Pond Dam downstream, and includes riverine, palustrine, and lacustrine wetland habitats. Immediately below the Allendale Dam, the river flows is channelized with a section of riffle/run habitat before the impact of the Lyman Mill Dam is encountered. Palustrine forested, scrub shrub, and emergent wetland habitats are associated with the riverine section, as well as the upper portion and borders of Lyman Mill Pond. The pond itself is classified as lacustrine wetland habitat based on its size (> 20 acres). Representative photographs of the Lyman Mill EA are provided in Appendix B.

The section of river below the Allendale dam is classified as upper perennial with a rock cobble bottom. The banks are steep and undercut for most of the section. The banks are vegetated with overhanging shrubs and trees including dogwood, witch hazel (*Hamamelis virginiana*), birch, red maples, and oaks (*Quercus* sp.), tree of heaven (*Ailanthus altissima*), catalpa (*Catalpa speciosa*), ash (*Fraxinus* sp.), and sweet gum (*Nyssa sylvatica*). A large forest borders the western bank, while a thin strip of scrub shrub habitat and residential properties as well as a ball field and recreational complex border the eastern bank. Several likely vernal pools were observed in the forested floodplain area downstream from the Allendale dam during the ecological field reconnaissance.

The riverine habitat transitions into a palustrine scrub shrub and emergent wetland at the upper end of Lyman Mill Pond. Button bush (*Cephalanthus occidentalis*) dominates the palustrine scrub shrub wetland north of Lyman Mill Pond. Subordinate species identified in this habitat included alders, dogwoods (*Cornus* sp.), purple loosestrife, poison ivy,

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cattails, and red maple saplings. Water willow dominates along the upper borders of the Lyman Mill Pond, between the palustrine and littoral lacustrine wetland habitat.

Lyman Mill Pond is characterized as a lacustrine wetland habitat. The majority of the pond is classified as littoral (i.e., less than 3 meters deep). Aquatic beds of water milfoil and pond weed (*Potamogeton* spp.) dominate the pond bottom and water column. Approximately 10 to 20 percent of the bottom is unvegetated; unvegetated areas are generally limited to confluences with tributaries (e.g., Assapumpset Brook) and immediately upstream of the Lyman Mill Dam. Osprey (*Pandion haliaetus*) were observed feeding frequently in Lyman Mill Pond in 2001 and fish observed in this reach include American eel, brown bullhead, tessellated darter, golden shiner (*Notemigonus crysoleucas*), bluegill (*Lepomis macrochirus*), chain pickerel, white sucker, and largemouth bass (*Micropterus salmoides*).

Manton. Habitat and river morphology is very similar to the Lyman Mill Pond Reach although the impounded portion of the river area behind the Manton Dam is approximately a tenth of the area (Figures 4 and 5). Although less information is available for this area (and Dyerville), it is likely that the flora and fauna associated with these EAs are similar to those described for upstream EAs. Representative photographs of the Manton EA are provided in Appendix B.

Dyerville. Only pilings remain of the former Dyerville Dam in the river and this lowest reach of the study area is primarily riverine in nature and classified as upper perennial with a rock cobble bottom. The banks are generally steep and undercut for most of the reach although scrub shrub habitat occurs along terraces. The banks are vegetated with overhanging shrubs and trees including dogwood, witch hazel, birch, red maple, oak, catalpa, and green ash (*Fraxinus pennsylvanica*). Representative photographs of the Dyerville EA are provided in Appendix B.

Assapumpset. This section is the offsite reference area for the Study Area. Assapumpset Pond is approximately 4 acres and primarily provides lacustrine wetland habitat although fringing scrub shrub wetland occurs along the western portion of the pond. The pond discharges to Assapumpset Brook at a spillway located to the southeast, which flows in a generally easterly direction before discharging to Lyman Mill Pond. The brook is narrow (approximately 3-4 feet wide) with a rocky bottom and is heavily shaded as it flows through upland forest habitat. Representative photographs of the Assapumpset reference area are provided in Appendix B.

2.1.1 Current and Reasonably Foreseeable Uses of the Site

The northern portion of the Site is occupied by the Brook Village and Centredale Manor apartment complexes. These parcels are currently occupied and covered by buildings, pavement, or landscaping. The remaining portions of the Site consist of reaches and

wetlands associated with the Woonasquatucket River. Per the Rhode Island Department of Environmental Management (RIDEM) Water Quality Regulations EVM 112-88.97-1 (June 23, 2000), the Woonasquatucket River (from Esmond Mill Drive in Smithfield to the combined sewer outfall (CSO) at Glenbridge Avenue in Providence) is classified as a Class B1 water body. Class B1 water bodies are:

> "designated for primary and secondary contact recreational activities and fish and wildlife habitat. They shall be suitable for compatible industrial processes and cooling, hydropower, aquacultural uses, navigation, and irrigation and other agricultural uses. These waters shall have good aesthetic value. Primary contact recreational activities may be impacted due to pathogens from approved wastewater discharges."

The land-use for the northern portion of the Site is expected to remain multi-family residential. USEPA plans to restore the Woonasquatucket River and associated reaches and ponds to a fishable and swimmable condition. The land-use for the eastern shore of the Allendale and Lyman reaches is primarily residential with some commercial and industrial activity. There are residential, commercial, and industrial properties approximately 200 feet or more from the western shore of the Allendale and Lyman Mill reaches. The majority of biological habitat associated with the floodplain of the Woonasquatucket River consists of wetlands, which are protected under both federal and state wetlands regulations, and available area should remain approximately the same in the future.

2.1.2 Summary of Environmental Conditions

Sampling activities conducted by USEPA and RIDEM revealed elevated polychlorinated dibenzo dioxins and furans (dioxins and furans) in soils and sediments as well as from fish taken from Woonasquatucket River. As mentioned above, other contaminants detected onsite include: PCBs, chlorinated and aromatic volatile organic compounds (VOCs), PAHs, phthalates, and metals. The Site was added to the National Priorities List on February 4, 2000. For the Woonasquatucket River, there is currently a fish consumption advisory in place that recommends that people not eat fish, eels, turtles, or plants from the river downstream of the Smithfield WWTP.

Approximately 400 drums and 6,000 cubic yards of soil were removed from the property by GZA GeoEnvironmental, Inc. during construction of the apartment complexes. However, the exact locations of these remediation activities are not known. Temporary caps were installed over heavily contaminated areas near residences. Two soil caps were placed in the source area in July 2000.

As part of time-critical removal activities conducted by USEPA, temporary fencing was erected around areas of contaminated surface soil in January 1999. The temporary

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fencing was replaced with chain-link fence between May and September 1999 to prevent access to contaminated areas.

General deterioration of the Allendale Dam resulted in it being partially breached in 1991, which allowed the water level in Allendale Pond to recede. Consequently, most of the pond bottom adjacent to residential properties along the eastern bank of Allendale Pond was exposed and much of the area became vegetated. USEPA planned to reconstruct Allendale Dam during the summer of 2001, thus restoring Allendale Pond. However, in May 2001, prior to any reconstruction efforts, two successive natural breaches occurred at Allendale Dam. The breaches in 1991 and 2001 likely resulted in the migration of contaminated sediment to Lyman Mill Pond in addition to a substantial lowering of Allendale Pond water. Since 2001, a new dam has been installed and water levels in Allendale Pond are back to levels that existed prior to the 1991 breach.

The residential soils and sediments along the eastern shore of Allendale Pond and Lyman Mill Pond have been the subject of previous investigations, and the need for remedial activities associated with those soils and sediments has been evaluated as part of the NTCRA. The details of the NTCRA objectives were identified in the *Request for Removal Action Centredale Manor Restoration Project, North Providence, Rhode Island-Action Memorandum – Non-Time-Critical Removal Action (NTCRA)* dated January 18, 2001 (USEPA, 2001a). In 2002, excavation of soils and sediment was conducted as part of the NTCRA. Based on the post excavation figures provided by USEPA, several environmental samples are no longer representative of Site conditions. The samples that were previously collected, but are no longer representative, have not been considered in this BERA. Table A-1 of Appendix A identifies all environmental samples utilized in the BERA. Table A-2 of Appendix A identifies soil and sediment sample locations that were not considered in the BERA for various reasons. Those samples removed due to NTCRA excavation activities are identified in Table A-2 as "Excavated" and "NTCRA". In addition, all soils classified as "residential" were not considered in this BERA.

Numerous soil sample locations no longer represent potential soil exposure opportunities for potential receptors at the Site. Soil sample locations within the footprint of the two caps (i.e., Cap #1 and Cap #2) and the rip-rap along the eastern bank of the Woonasquatucket River adjacent to and downstream of the source area are no longer representative of soils with current exposure potential. The caps, rip-rap, and existing paved areas will be evaluated as part of the permanent remedy for the source area. These contaminated soil areas and a need for caps were evaluated as part of the EPA Region I Action memorandum, dated May 4, 1999, as amended September 13, 1999 and June 1, 2000, and were addressed under the Time-Critical Removal Action (TCRA) in 2000. Therefore, soil samples from those areas have not been considered in this risk assessment. Soil sample locations from these areas that are not considered in the BERA are identified in Table A-2 as "Allendale" and either "Cap #1", "Cap #2", or "Rip-Rap.

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In addition, the drainage swale area located to the east of Centredale Manor is the focus of planned remedial activities (Battelle, 2003). The storm water that typically flows through this area will be diverted and the area will be covered with a two-foot thick cap. Therefore, samples collected from within this area will not be representative of potential soil or sediment exposure in the future and are not considered in this BERA. Table A-2 identifies those sample locations as "Allendale" and "2 ft Cap – Sluiceway".

As indicated above, due to the breaches of the Allendale Dam (in 1991 and in May 2001), water levels within Allendale reach have fluctuated, and have generally been lower than they were prior to 1991. Many of the soil samples collected from the Allendale Reach were collected at a time when the water levels were uncharacteristically low due to the breach of the Allendale Dam. The Allendale Dam was restored in 2002. As indicated in the Engineering Evaluation/Cost Analysis (EE/CA) (Tetra-Tech, 2000), the water level in Allendale with the dam restored would be 93.5 feet above sea level. Therefore, the typical water level in the Allendale Reach is expected to be 93.5 feet. Numerous soil samples have been identified that were collected as "soil" samples that are now, based on a water elevation of 93.5 feet, actually sediments. Figure 3-1 of the EE/CA was used to identify the extent of area within Allendale Reach that is below elevation 93.5. All "soil" samples collected from that area have been treated as "sediment" for the purposes of the BERA because the locations of those samples are typically below water. Table A-2 in Appendix A documents those "soil" samples that are now "sediments" because the water level in Allendale has been restored to historical levels.

2.2 CONCEPTUAL SITE MODEL

The CSM identifies potential source areas from which COPCs may have been released, the migration pathways through which they may have been transported and/or translocated to other environmental media, and where possible exposure may occur. The CSM provides a framework for understanding contaminant sources, migration pathways, identification of potential receptors, and development of exposure profiles.

2.2.1 Source Area

The main area of the Site, consisting of approximately 9.04 acres, is located in North Providence, Rhode Island, just south of Route 44 on the eastern bank of the Woonasquatucket River (Figure 1). The Site (source area) consists of two parcels located at 2072 and 2074 Smith Street (Lots 200 & 250) that cover approximately 9.04 acres (see Figure 2). The remaining portions of the Site consist of reaches, man-made ponds, and wetlands associated with the Woonasquatucket River. The river flows from north to south (Figure 2). A drainage swale extends south from the northeast portion of the Site, curves to the west and discharges into the Woonasquatucket River and Allendale Pond south of the Site.

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The Site was used for disposal of wastes containing hazardous substances. Between approximately 1940 and 1970, USEPA has reason to believe that the main area of the Site was the location of a chemical manufacturing facility (which produced hexachlorophene, among other chemicals) and an incineration-based drum recycling facility. Evidence suggests that operations at the former chemical company and drum reconditioning facility resulted in waste disposal onto surface soil and beneath the ground surface. Wastes have also been released directly into the Woonasquatucket River, which runs along the western side of the source area (Tetra Tech NUS Inc., 2000).

Currently, two high rise buildings (Centredale Manor and Brook Village, both federally-subsidized, senior housing complexes) are located in the main area of the Site, which is zoned for residential occupancy. In addition to the buildings, the main area is covered by roadway, parking lots, and the two capped source areas. The first housing complex was built in approximately 1976. The second high rise was built in 1982. Construction records show that hazardous substances were removed from the Site during the construction of the second complex. Samples indicate the presence of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and other contaminants in soil and sediment at the main portion of the Site and the river sediment and floodplain soils downstream. Dioxins and furans have been detected in soils and sediments as well as in fish tissue collected in 1996 from the Woonasquatucket River. Other contaminants detected in Site media include PCBs, VOCs, PAHs, HCX, phthalates, and metals. Elevated VOCs have been detected in overburden groundwater at the Site and are discharging to the adjacent reach of the Woonasquatucket River as confirmed by vapor diffusion samplers (Church et al., 2000).

2.2.2 Migration of Contaminants

Downstream sediments have been impacted through the transport and deposition of contaminants from the source area. Dioxins and furans have been detected in sediments at the Allendale, Lyman Mill and Manton EAs (Figure 2). With the partial breaching of Allendale Dam in 1991, and the more recent breach in 2001, further contaminant migration may have occurred.

Contaminants that were discharged from the source area entered the Woonasquatucket River and were transported downstream either dissolved in the water column or adsorbed to resuspended sediments. Overland flow during flooding events resulted in contamination of lateral floodplain habitats associated with the river. Over time, contaminants preferentially accumulated in low energy sediment and floodplain soils, which are characterized by higher organic carbon content. Compounds with a propensity to bioaccumulate were taken up by plants, invertebrates, and fish and were transferred through aquatic food webs. Wildlife species that consume these lower trophic level organisms could also be exposed to site-related contaminants. Contaminants that were

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deposited in floodplain soils could also enter the terrestrial food webs by a similar process.

2.2.3 Potential Ecological Receptors

In general, aquatic receptors (including invertebrates and both demersal and pelagic fish species) are exposed to COPCs in sediment and surface water via direct contact, direct ingestion, or by consuming prey items that have bioaccumulated COPCs. Semi-aquatic receptors (including mammals, birds, reptiles, and amphibians) may be exposed as a result of incidental ingestion of sediment, consumption of water, or ingestion of contaminated prey. Terrestrial invertebrates and wildlife that prey on these species may be exposed to COPCs in floodplain soil directly or by ingesting contaminated prey. The Work Plan (Harding ESE, 2001a) describes the receptors and exposure pathways that were selected for quantitative evaluation.

Potential ecological receptor species considered in this BERA are aquatic and floodplain invertebrates, amphibians, fish, birds, and mammals that depend on aquatic resources of the Woonasquatucket River. This section provides a general description of the ecological receptors that reside in or around the Woonasquatucket River as well as a brief discussion of their ecological role and importance. Life history details for selected receptors area also provided. Figure 6 provides a diagrammatic depiction of the important exposure pathways for each of these major ecological receptor groups.

2.2.3.1 Aquatic Macroinvertebrate Taxa

Invertebrate communities constitute a significant basis of the food chain in aquatic ecosystems. Since invertebrates process organic material and are the prey items of other invertebrates, fish, birds and mammals, they are important in nutrient and energy transfer in an aquatic ecosystem. Alterations of invertebrate function may consequently affect nutrient and energy transfer and populations of organisms that prey upon them. In addition, COPCs in invertebrates may be passed along through the food chain. Therefore, upper trophic levels can be affected not only by reduced prey abundance, but also by trophic transfer of accumulated contaminants in invertebrate prey.

Water column invertebrates include the general class of zooplankton common in all freshwater bodies. These organisms are extremely important in nutrient and energy transfer and are a critical resource to juvenile fish and macroinvertebrate larvae and nymphs. Benthic invertebrate communities are a heterogeneous assemblage of organisms that inhabit bottom substrates and, like pelagic invertebrates, constitute a substantial basis of the aquatic food chain. Sediment-associated macroinvertebrates include larval and nymphal insects, amphipods, isopods, and crayfish. Benthic invertebrates are susceptible to COPC exposure because they live and feed directly in the sediment, where most contaminants come to reside. Benthic macroinvertebrates play several important roles in

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the aquatic community including mineralization and recycling of organic matter and, therefore, nutrient and energy cycling which supports the productivity of the entire ecosystem. Also, benthic invertebrates are important trophic links in aquatic communities because they consume bacteria, plankton, and detritus, and are a dominant prey base for certain species of fish, birds, mammals, and other benthic organisms.

Characteristic of larger rivers in industrialized New England, the benthic and epibenthic community in riffles of the study area are dominated by several families within Trichoptera (caddis flies) and Diptera (midges) (Appendix E). The soft depositional zones tend to be dominated by oligochaete worms (*Nais* sp.). Important epibenthic crustaceans found in the study area are the crayfish species *Orconectes limosus* and *Procambarus acutus*.

Although a formal census of the freshwater mussel fauna occurring within the Woonasquatucket River was not conducted, it is likely that the available habitat could support representative unionid and fingernail clams. A total of 33 species of freshwater mussels and clams have been described occurring in the freshwater drainages of southern New England common species found in rivers and slow moving waters in Rhode Island include *Elliptio complanata*, *Pyganodon cataracta*, *Musculium securis*, *Lampsilis radiata*, and *Sphaerium simile* (Rice, 2004).

Crayfish. Crayfish are large macroinvertebrates in the Order Decapoda found in most freshwater habitats worldwide although the U.S. fauna is restricted to North America. Adults hide in burrows or under rocks and submerged logs throughout the day and forage at night, generally in water depths less than 3 feet (Pennak, 1978). Although crayfish are known to migrate seasonally into deeper water in lacustrine habitat, in streams most individuals are relatively sedentary with home ranges less than 100 feet. Population densities vary considerably depending on the species and habitat but 20,000 individuals/acre is considered typical of pond environments, with somewhat lower densities occurring in streams (Pennak, 1978).

Crayfish are omnivorous but seldom take live prey, preferring to scavenge detritus (both plant and animal). The normal life span is generally less than 2 years, with individuals sexually mature within a year of birth. Females lay their eggs weeks to months after copulation and sperm transfer at which time she positions the fertilized eggs on short stalks to her abdomen (i.e., "in berry"). The eggs are incubated for periods ranging up to 20 weeks and kept aerated by movement of the female's pleopods (Pennak, 1978). The young undergo a series of molts before attaining adult size.

2.2.3.2 Floodplain Soil Macroinvertebrate Taxa

The floodplain soil invertebrate community consists of various insect, oligochaete, collembola, mite taxa, among others. As with aquatic macroinvertebrates, floodplain soil

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invertebrates play a critical role in energy and nutrient transfer. Direct effects of COPCs on soil invertebrates can reduce the productivity of soil and impact the populations of animals that rely on these organisms for forage.

Soil invertebrates play an important role in nutrient and energy cycling and serve as part of the forage base for terrestrial and semi-aquatic wildlife. A survey of the soil invertebrates associated with study area floodplain soils conducted in 2001 (Appendix E) identified an earthworm dominated (particularly *Lumbricus rubellus* and *Aporrectodea rosea*) macroinvertebrate community fauna. Other represented invertebrate groups include both adult and/or immature insect taxa (including beetles, moths, flies, bugs, and ants), spiders, sow bugs, slugs, centipedes, millipedes, and nematode worms (Appendix E).

2.2.3.3 Amphibians/Reptiles

An anuran call survey was conducted during the amphibian breeding season in both 2001 and 2002 (USFWS, 2001; 2002). A diverse assemblage of amphibians was identified, with the following species identified in one or more locations within the study area: wood frog (*Rana sylvatica*), pickerel frog (*R. palustris*), green frog, bullfrog (*R. catesbeiana*), spring peeper (*Pseudacris crucifer*), gray treefrog (*Hyla versicolor*), American toad (*Bufo americanus*), and Fowler's toad (*B. woodhousei fowleri*).

2.2.3.4 Fish

Demersal fish live in contact with and forage for food directly in the sediments. As such, they represent a unique exposure pathway because of their foraging behavior (i.e., substantial and direct exposure to sediments) and prey items (i.e., predominantly benthic invertebrates). Impairment to benthic fish communities could have substantial impacts on nutrient and energy cycling, and on instream and nearby upland biological communities. Examples of common demersal fish in the Woonasquatucket River include white sucker and brown bullhead. The American eel is also closely associated with the sediment substrate.

Pelagic fish live and feed principally in the water column (as opposed to being in direct contact with sediment). Pelagic fish represent many trophic levels with prey items predominately in the water column (e.g., zooplankton and other fish). Upper trophic level pelagic fish may be strongly impacted by food chain transfer of COPCs. As with benthic fish, several pelagic fish are of commercial and/or recreational importance and a decline in these fish species could have an economic impact on the region. Examples of important pelagic fish that are known to occur in the Woonasquatucket River include bluegill sunfish, redfin pickerel (*Esox americanus*), chain pickerel, and largemouth bass. In addition, the Woonasquatucket River is a designated trout water, with rainbow trout (*Oncorhynchus mykiss*) stocked yearly at Smithfield (RIDEM, 2003).

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White Sucker. White sucker is an abundant species that occurs from Nova Scotia, south to Georgia and west to British Columbia and the Mackenzie River delta (Scott and Crossman, 1973). Adult white sucker are a benthic dwelling, benthic feeding species that generally lives and feeds in direct contact with the sediments (Scott and Crossman, 1973). The small downward point mouth and large protruding lips make the species uniquely adapted to vacuum feed on the river bottom, ingesting amphipods, larval aquatic insects, clams, snails, and plant material. One study found the percentage composition of the gut of individual species to include 5-90% midge larvae, 2-70% caddisflies, and 5-85% mollusks (Scott and Crossman, 1973). Detritus can also be a component of the diet, but the quantity in white sucker foreguts was inversely related to benthic microcrustacean densities. The species can apparently separate detritus from invertebrates, suggesting that detritus is not ingested incidentally, but intentionally when preferred prey is scarce (Ahlgren, 1990).

Home ranges of white sucker have not been well studied. In some watersheds, the species has displayed distinct spawning migrations from lake to tributaries in the spring. In others, there do not appear to be large movements. The species can spawn in both streams and lakes (Scott and Crossman, 1973).

Brown Bullhead. The brown bullhead is native to freshwater habitats throughout eastern and central North America and generally are found at or near the bottom of shallow warm water ponds, lakes, and slow-moving streams and rivers (Scott and Crossman, 1973). Preferred habitat provides abundant aquatic vegetation and sand to muck bottom substrates. The brown bullhead is considered relatively tolerant to elevated temperature and low dissolved oxygen conditions that may be limiting to other species. Adult fish are omnivorous, feed on a wide variety of aquatic prey including insects, crustaceans, detritus, mollusks, worms, fish, and decaying animal and plant detritus (Scott and Crossman, 1973). They are also known to be significant consumers of fish eggs. Bullheads possess distinctive barbels (sensory organs) that are used to locate prey during their nocturnal foraging and to navigate in low visibility habitat. The young feed on midge larvae, amphipods, and various pelagic microcrustaceans. This species is consumed by various piscivorous fish (e.g., northern pike, chain pickerel, walleye) but the presence of substantial pectoral spines in larger sized individuals serves as a predator deterrent.

The brown bullhead spawns in late spring and summer and may spawn more than once each season. Eggs are deposited in shallow nests, nest burrows or artificial substrates and parental care continues for several weeks after egg hatch (Scott and Crossman, 1973). The young grow fairly rapidly to adult size of approximately 0.75 – 1 pound and attain sexual maturity in 3 years; the average lifespan is believed to be 6-8 years.

Largemouth Bass. The largemouth bass occurs throughout the entire Atlantic coast (rarely into brackish water) from Maine to Florida, west to Texas, north to North Dakota and east along southern Canada (Scott and Crossman, 1973). Preferred habitat for largemouth bass is shallow warm water lakes and ponds, and larger, slow-moving rivers; it is almost universally found in association with soft bottom substrate, stumps, and extensive growths of a variety of emergent and submergent aquatic vegetation (especially water lilies, cattails, and pond weed) (Scott and Crossman, 1973). Largemouth bass exhibit strong habitat fidelity, with limited seasonal movement and small territory size. Adult fish are primarily piscivorous but young fish, after feeding initially on plankton, will consume primarily invertebrates and amphibians. Feeding occurs throughout the day and into evening, and occurs in schools along the shore usually near vegetation.

The largemouth bass spawns from late spring to mid-summer (peaking in early to mid-June). Males build nests, typically in soft mud associated with emergent vegetation, aggressively defend the spawning area, and then guard and aerate the fertilized eggs. Fry hatch within a week but remain in the brood nest for up to a month during which time growth is rapid (Scott and Crossman, 1973). Sexual maturity is reached in 4 to 5 years and most individuals do not attain a size greater than several pounds.

American Eel. The American eel occurs from southwest Greenland south along the Atlantic coast into the Caribbean to Panama (Scott and Crossman, 1973). It is found in coastal areas and freshwaters inland to the Great Lakes and up the Mississippi River. The American eel is a catadromous species and in the autumn mature adults migrate from freshwater habitat into the ocean to spawn in the Sargasso Sea east of the Bahamas. Young eels require a year to grow and migrate into freshwater streams and rivers that discharge along the Atlantic coastline; migration of the now pigmented animals (called elvers) into freshwater occurs in the spring.

Eels are voracious nocturnal predators consuming a wide variety of fish and invertebrate prey. In one study, 90% of the diet consisted of insect larvae and nymphs (mayfly, dragonfly), crayfish, and snails (Scott and Crossman, 1973). Eels move freely into muddy depositional areas of lake where they lay hidden during the daytime. Eels are also are able to scale nearly vertical walls of dams and impoundments and even travel short distances overland during their migration into freshwaters. Due to their agility in climbing dams, the American eel is the only species that is able to traverse the multiple EAs.

2.2.3.5 Wildlife

All primary wildlife feeding guilds are represented within the study area including piscivores, which feed primarily on fish; insectivores, which feed primarily on insects (this guild includes species that specialized on earthworms [i.e., vermivores]); omnivores, which feed on a variety of prey types; and carnivores. Herbivorous wildlife were not

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evaluated specifically in the BERA because these receptors are unlikely to be as substantially exposed as those that feed at higher trophic levels.

Piscivorous birds and mammals rely primarily on fish as food, but may also consume amphibians, invertebrates, crayfish, clams, and mussels. These species feed at the top of the aquatic food chain and may therefore receive higher contaminant exposures to bioaccumulating contaminants than do representatives of other trophic levels. Examples of piscivorous birds found in the Woonasquatucket River watershed include the belted kingfisher, great blue heron, and osprey. In 2000, USEPA observed a colony (approximately 6 individuals) of great egrets (*Casmerodius albus*) nesting in tree tops along the western shore of Allendale Pond. The otter (*Lutra canadensis*) is a piscivorous mammal that has historically been reported occurring within the Woonasquatucket River watershed.

Insectivorous birds and mammals forage primarily on insects, which are captured in the air or are gleaned off vegetation and other substrates. Aerial foraging species include tree swallows and the little brown bat (*Myotis lucifugus*); surface gleaners are well represented by various warbler species. Wildlife that forages predominately on earthworms found in floodplain soils includes woodcock (*Scolopax minor*) and the short-tailed shrew (*Blarina brevicauda).* The direct association between benthic invertebrates and potentially contaminated sediment, and earthworms with potentially contaminated soils can result in increased exposure to insectivorous species that forage on emerging aquatic insects and vermivorous species that consume earthworms.

Omnivorous species feed on insects, small mammals, birds, reptiles, mussels, and fruit. Omnivorous species are typically opportunistic hunters, feeding on whatever is seasonally or locally abundant. Dietary shifts are common and often linked to the need for protein-rich foods in preparation for breeding. Their varied dietary preferences and inclusion of both aquatic- and terrestrial-prey items in their diet expose them to contaminants that bioaccumulated in different prey items. The raccoon is a widespread species that represents this trophic category at the Site. Omnivorous birds such as the mallard, feed predominantly on plant material although invertebrates are also preyed upon (particularly in preparation for breeding and egg production). The sediment filtering habit of surface feeding ducks exposes them directly to sediment contaminants.

Carnivorous birds and mammals consume primarily animal tissue and representatives of this foraging guild that likely forage in the Woonasquatucket River include red fox (*Vulpes*), red-tailed hawk (*Buteo jamaicensis*) and various hawks of the genus Accipter (accipiters).

Mallard. The mallard is a permanent resident of the southern New England states and the breeding range includes the northern half of the U.S., south to Virginia, Kansas and northern New Mexico (DeGraaf and Rudis, 1983). It is a commonly observed species

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found in most ponds, lakes, rivers, streams, marshes, wet meadows, and wooded swamps. Preferred habitat provides shallow water (less than 16 inches deep) that allows the duck to feed on submerged vegetation by dabbling. Home range is about 2 square miles during breeding season and densities of 5 to 6 pairs per acre are not uncommon (DeGraaf and Rudis, 1983). Preferred forage of the mallard consists of seeds of various wetland plants supplemented by leafs and stems of aquatic plants, snails, and insects.

Nests typically are found on the ground near the edge of a water body. Females lay one brood of between 6 to 15 eggs per year between mid spring and the end of June. Eggs are incubated for about a month and the precocial chicks are out of the nest within a day (DeGraaf and Rudis, 1983).

Belted Kingfisher. The belted kingfisher is a common piscivorous bird with a breeding range throughout the United States and much of Canada; it is a permanent resident in southern New England (DeGraaf and Rudis, 1983). The kingfisher breeds on banks near ponds, lakes, rivers, and streams, and forages by diving for prey in shallow water sighted from perches. Fish is the staple component of the kingfisher diet but crayfish, insects, mollusks, and tadpoles are also consumed. A clutch, consisting of between 5 to 8 eggs, is laid in May and incubated for a little over 3 weeks; the chicks remain in the nest for about a month (DeGraaf and Rudis, 1983). Birds are sexually mature the year following their birth.

Great Blue Heron. The great blue heron is a semi-aquatic wading bird that breeds throughout the United States and winters as far north as New England and southern Alaska. Spring migrants return to Illinois, Wisconsin, and central Minnesota in early February and continue moving into the U.S. and Canada throughout May. Great blue herons are generally inclined to return to the same breeding area each year. The great blue heron favors natural wetlands and riverbanks, but will nest on ocean shores, in brackish marshes and lagoons, and on lakes. These birds nest mostly in colonies of several hundred pairs and prefer islands or wooded swamps where they are isolated from predators; however, ground, cliffs, reeds or rushes may also be used.

Great blue herons feed by wading slowly in waters along shorelines to catch fish and invertebrate prey. They hunt by sight both night and day and will often stand and wait for prey to approach them. Herons often feed in groups unless resources are limited (Kulshan, 1978). Because great blue heron are primarily piscivores, their abundance in a particular habitat is positively influenced by the presence of small fish in shallow streams. The species of fish a heron chooses as prey depends on the size of the fish and the habitat in which the heron is foraging. The size of fish caught by herons has been shown to vary with habitat. Herons foraging in streams and rivers tend to catch a larger size range of fish, 8-23 cm and 8-33 cm, respectively, while heron foraging in lakes tend to capture fish from 20 to 28 cm in length (Alexander, 1977). Amphibians, invertebrates, reptiles, small mammals, and other birds may also be present in the diet of the great blue

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heron, especially in winter when the herons may move upland and forage in wet meadows and pastures (Butler, 1992; Kulshan, 1978). Small mammals, such as rodents, can play a large role in the diet of the great blue heron in early spring when the heron has just returned from the breeding season.

Females reach sexual maturity during their second year. Clutches (between 3 to 7 eggs) are set during the spring and early summer and incubated for about a month (DeGraaf and Rudis, 1983). Chicks remain in the nest for 2 months.

Tree Swallow. The tree swallow is a common and abundant insectivorous bird that breeds from throughout central and northern North America, with localized populations in the west; the swallow flies south as far as Mexico and Central America to winter. Males, and to a lesser extent females, exhibit strong breeding site fidelity (Robertson et al., 1992). During the breeding season, swallows occur in river bottomlands, beaver ponds, wooded swamps and marshes where dead standing trees are found. The availability of cavities for nesting is an essential habitat requirement that may limit the population densities of tree swallows; densities of over 200 pairs per acre are possible when nest boxes are deployed (DeGraaf and Rudis, 1983). Flying insects are the staple component of the tree swallow diet but berries and seeds will be consumed when insects are not abundant. Foraging typically occurs over open bodies of water with individuals skimming over the surface to capture prey.

Eggs are laid in May and June in clutches of between 4 to 7 eggs, which are incubated for approximately 2 weeks; the altricial nestlings remain in the nest for between 16 to 24 days depending on the abundance of food (DeGraaf and Rudis, 1983). Adults attain sexual maturity the year after being born.

American Woodcock. The American woodcock breeds from southern Canada south to Texas and Florida and the winter range extends north to southern New Jersey and the Ohio Valley (DeGraaf and Rudis, 1983). In New England, the woodcock is most common in Maine and is uncommon in the southern states. It requires moist woodlands in early stages of succession, swamps, stream banks, or rich bottomlands that support abundant earthworm populations; roosting and spring courtship activities occur in fields or small forest openings. Earthworms are the dietary staple (ranging from 50 to 90 percent) of woodcocks but individuals also consume insect larvae. The long woodcock bill is used to probe or glean food from the substrate. Optimal habitat conditions support one pair of birds per 6 acres (DeGraaf and Rudis, 1983).

Eggs are laid from late spring to the middle of June with a typical clutch containing 4 eggs which are incubated for approximately 3 weeks. Chicks are precocial and leave the nest within several days after hatching and reach sexual maturity within a year.

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Little Brown Bat. The little brown bat occurs from Labrador to southern Alaska, south to California and in the Appalachians to Georgia (DeGraaf and Rudis, 1983). It is known to be one of the most abundant of all bat species in the northern part of its range (Godin, 1977; DeGraaf and Rudis, 1983) and in the spring and summer females form maternity colonies of hundreds in various retreats including attics and barns, typically near streams or lakes (Baker, 1983). In New England, females end hibernation from early April to mid-May and scatter to summer colonies moving as far as 170 miles from the hibernaculum (Godin, 1977). Individuals return to the winter roost for hibernation in September and October. The local summer population size appears to be correlated with distance from the winter hibernaculum (Baker, 1983).

Little brown bats take wing shortly after dusk and seek water, where they skim the surface to drink and capture insects. Individuals have voracious appetites feeding on flying insects, particularly nocturnal moths, bugs, beetles, flies and mosquitoes, which are often caught with the wing or tail membrane. The little brown bat makes several foraging bouts during the night and can consume a fifth of their body weight within an hour span (Godin, 1977). Food transit time through the body for an active individual is as short as 35 minutes.

Although courtship and mating in little brown bats occurs in September, ovulation is delayed until the following spring and young are born following a 2 month gestation period. Young bats grow rapidly and are able to fly within 3 weeks of birth (Baker, 1983). Females may attain sexual maturity during the year of their birth and life expectancy is surprisingly long, with individuals known to live in nature for up to 30 years.

River Otter. The otter is distributed widely throughout North America with the exception of the northernmost portions of Canada and Alaska (DeGraaf and Rudis, 1983) and is considered to be relatively uncommon throughout much of its range although it is known to occur within the Woonasquatucket River watershed. The otter is found along borders of streams and wetlands and lake shorelines in forested areas; in Michigan, the otter is commonly associated with shallow emergent marshes (Baker, 1983). The literature reports that the home range of male and pairs can range up to 20 to 30 linear miles of stream or river (DeGraaf and Rudis, 1983) but is usually less than a mile for females with young. These low densities are believed to be due to the mobility, food requirements, and territoriality of these animals; however population censuses may often underestimate densities because up to 70 percent of individuals are either non-territory holding young-of-the-year or transients (Baker, 1983). Otter consume a variety of aquatic animals including fish, amphibians, crayfish, and turtles and will opportunistically feed on small birds and mammals, earthworms, and snakes. The fish taken by otters is usually the slower swimming forage fish rather than the faster top predators (Baker, 1983). Individuals are active throughout the day and during all seasons. Rock crevices,

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abandoned beaver or muskrat lodges or dense thickets provide denning habitat for the otter (DeGraaf and Rudis, 1983).

Adults reach sexual maturity in 2 to 3 years and breeding occurs usually in early to mid-spring followed by a 10 to 11 month delayed implantation period. The average 2-3 fetuses in a litter are gestated for up to a year and are born in March or April. In nature, otters live as long as 8 to 13 years. The combination of relatively small litters and length of time for females to reach sexual maturity results in a low recruitment rate for this species (Baker, 1983).

Short-tailed Shrew. The short-tailed shrew occurs from Nova Scotia, west to Sasketchewan, south to East Texas and southern Florida and is common in New England (DeGraaf and Rudis, 1983). The short-tailed shrew occurs in both forested and successional habitats but is especially common along streams in scrub/shrub habitat characterized by loose leaf litter and high humidity. In optimal habitat, population estimates range as high as 50 individuals per acre (Baker, 1983). This species is a voracious nocturnal and diurnal voracious feeder consuming insects, plants worms, sowbugs, snails and even small vertebrates. Adult home range size is approximately an acre and good habitat can support up to 50 individuals per acre (DeGraaf and Rudis, 1983).

Females breed within a year of birth and may have 2 to 3 litters throughout the spring and summer months (DeGraaf and Rudis, 1983). Litter size ranges from 3 to 10 and pups are born between April and September following a 3 week gestation period.

Raccoon. The raccoon is a common mid-sized mammal that occurs throughout most of southern Canada and the United States. It is primarily a nocturnal animal that is dormant (but not in hibernation) during the winter. Preferred natural habitat consists of a mosaic of woods, fields and abundant water resources; hollow logs are preferred for denning (DeGraaf and Rudis, 1983). Home range size varies with habitat quality and weather but is generally about a mile. Hollow logs, trees, rock crevices, ground burrows, or barns and abandoned buildings are utilized for den sites. Population densities can range from 1 to 15 raccoons per acre, depending on the habitat quality. The raccoon is an omnivorous and opportunistic feeder consuming primarily animal prey in the spring and early summer and switching to available fruits and seeds from late summer through the winter. Typical animal prey includes crayfish, worms, insects and carrion.

Females reach sexual maturity in either their first or second year and breeding occurs in mid winter to early spring. Following a 2 month gestation period, between 3 to 7 cubs are born in late April to early May (DeGraaf and Rudis, 1983). The typical lifespan is about 10 to 12 years.

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2.2.3.6 Rare, Threatened and Endangered Receptors

The U.S. Fish and Wildlife Service (USFWS) and RIDEM were contacted regarding the presence of rare, threatened, or endangered species known to occur in the vicinity of the study area. The USFWS (2002) indicated that no rare, threatened, or endangered species are known to occur in the area of the Site, with the possible exception of transient bald eagles (*Haliaeetus leucocephalus*). The USFWS and RIDEM responses are provided in Appendix C. The osprey and great blue heron are state-listed species of concern.

2.2.4 Complete Exposure Pathways

The conceptual model identifies the source, media, pathway and route of exposures evaluated in the ecological risk assessment, and the relationship between the measurement endpoints to the assessment endpoints (USEPA, 1997). It serves as a communication tool that illustrates the major pathways by which ecological receptors might be exposed to COPCs associated with releases from the Centredale Manor Site source area. Figure 3 presents a generalized site conceptual model for the site.

Ecological receptors may be exposed to site-related contaminants via a variety of exposure pathways. A complete exposure pathway involves a potential for contact between a given receptor and contamination either through direct exposure to an abiotic medium or indirectly through prey consumption. Pathways are evaluated by considering information on contaminant fate and transport, ecosystems potentially affected, and the magnitude and extent of contamination (USEPA, 1997).

Animals and plants that occur in or adjacent to the Woonasquatucket River, including invertebrates, fish, amphibians, birds, and mammals could be exposed to contaminants through contact with floodplain soil, sediment, surface water, and prey consumption. Species representing various trophic levels were selected as representative receptor species to evaluate the assessment endpoints discussed in Section 2.6. The selected species are intended to be representative of other species at the same trophic level that share similar ecological characteristics. These groups of species are generally referred to as guilds. By evaluating a representative member of a guild and by accounting for the predominant guilds, the uncertainty associated with missing an important species group or pathway is reduced. The following section describes the pathways by which the representative receptor species or taxa could be exposed to contaminants that have come to reside within the study area.

2.2.4.1 Aquatic Exposure Pathways

Aquatic and semi-aquatic organisms, such as plants, invertebrates, amphibians, and fish are exposed to contaminants through direct uptake from water, uptake from sediment,

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and/or uptake via food. Exposure is dependent upon timing (e.g., life-stage), feeding preferences, and length of time of exposure.

Organisms exposed to contaminants primarily via the water column include lower trophic level pelagic or planktonic species that live suspended or swimming in the water column. Uptake from sediment is dependent on a number of factors including contaminant and organic carbon concentrations. Habitat selection of aquatic organisms plays a role in the potential exposure to sediment contaminants. Organisms that prefer fine-grained sediments may be exposed to higher contaminant concentrations. Direct contact with and or ingestion of contaminated sediment and associated pore water are the primary routes of exposure for benthic infauna that live in close association with or are buried in the sediment. Epifaunal organisms living on the surface of the sediment receive exposure from both the sediment and overlying water. Uptake via food consumption will generally be the primary exposure pathway for bioaccumulating compounds.

Although receptors such as tree swallows and bats are not considered aquatic, their primary prey items are intimately connected to sediments and these receptors can be exposed to compounds that are accumulated in juvenile invertebrates when they become adults and take to the air to reproduce. The foraging and care provisioning by adult insectivorous birds and mammals can result in exposure to juveniles while the consumption of contaminated prey by the adults themselves can result in embryo exposures.

2.2.4.2 Floodplain Exposure Pathways

Terrestrial animals, such as invertebrates, amphibians, birds, and mammals may be exposed to contaminants that have migrated to floodplain soils. Potential exposure pathways include food intake, surface water ingestion, incidental soil ingestion, dermal contact, and under certain circumstances, via inhalation. The dermal absorption and inhalation exposure routes were not evaluated in this BERA due to the anticipated insignificance of these pathways, as well as substantial estimation uncertainties.

2.3 SUMMARY OF DATA

Harding ESE previously identified project goals, objectives, and data requirements to support human health and ecological risk assessments for the Site (Harding ESE, 2000). In the Work Plan prepared to support the human health and ecological risk assessments (Harding ESE, 2001a), Harding ESE evaluated the existing data and documents (as of March 2001) provided by USEPA and the USACE. Available information was reviewed to determine the need for additional data for successfully accomplishing the planned biota consumption and ecological risk assessments. During the data review, the number of samples collected for each medium, chemical analyses performed for samples collected during particular sampling events, and the location of samples collected were evaluated.

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It was determined that insufficient historical biota, soil, sediment, and surface water were available to perform the BHHRA and BERA and that additional samples needed to be collected from these media to fill the data gaps. Based on this information, a Field Sampling Plan (FSP) (Harding ESE, 2001b) was prepared to identify the specific data that would be required for each medium in each exposure area to evaluate the human health biota consumption pathway and ecological exposure pathways.

The BHHRA/BERA 2001 field collection activities began in May 2001. Tissue samples collected in the Allendale and Lyman Mill reaches and in upstream areas included fish, crayfish, emerging insects, and earthworms. Media sampled in those areas included floodplain soil, aquatic sediment, and surface water. In the Assapumpset reference area, fish, crayfish, aquatic sediment, and surface water samples were also collected. Fish sampling was conducted in Manton and Dyerville Ponds. No other sampling occurred in those reaches. BHHRA/BERA field activities were completed in late July 2001. The abiotic media data were collected to support the bioassessment data. A variety of bioassessment studies were identified as necessary to conduct the ecological risk assessment (e.g., floodplain invertebrate study, aquatic macroinvertebrate study). The additional sediment, soil, and surface water samples were necessary to interpret the additional biological data.

The samples collected during the BHHRA/BERA field sampling program were analyzed or tested for chemical, physical, and biological parameters by laboratories in several organizations, including:

Parameters	Laboratory
Dioxin/Furan and PCB Congener	Battelle, Columbus OH
HCX	Battelle, Columbus OH
TCX	Battelle, Columbus OH
PCB Aroclor and Chlorinated Pesticides	Battelle, Duxbury MA
Polycyclic aromatic hydrocarbons	Battelle, Duxbury MA
Metals, AVS/SEM	Battelle, Sequim WA
SVOCs and VOCs	Severn Trent Laboratories, Baltimore MD
BOD	Cassell Testing, Hunt Valley MD
Nutrients	Chesapeake Biological Laboratory, Solomons MD
Sediment Toxicity	Great Lakes Environmental Center, Traverse City MI
Taxonomy	Normandeau Associates, Bedford NH

Additional field efforts and analyses were performed by other parties in 2001. USGS collected tree swallow samples in May and June of 2000 through 2003. USACE

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collected fish data to support the ELS testing and the USFWS conducted an anuran call survey in 2000 and 2001. Battelle performed the ELS bioassay study. TetraTech collected floodplain soil samples in June and July 2001, sediment samples in July 2001, and groundwater samples in August 2001.

In the Final Data Evaluation Report (DER) (MACTEC, 2003) historical analytical data, including sediment, surface water, floodplain and bank soil, and biota were compiled and reviewed. The DER also presents and summarizes the biota, sediment, surface water, and soils data collected during the 2001 Spring and Summer BHHRA/BERA investigation activities in and adjacent to the Woonasquatucket River and at Assapumpset Brook and Pond.

The historical data, data from the 2001 BHHRA/BERA investigation, and data collected to support ecological risk assessment activities have been considered for use in this BERA.

In addition to the data summaries provided in the DER, a graphical presentation of data may be useful in identifying potential EAs that exhibit analytical parameters that have elevated frequency of detection and/or elevated concentrations relative to conditions at background and reference areas that are expected to be free of potential impacts from the Site. The upgradient background and reference areas considered in the BERA, Greystone and Assapumpset Brook/Pond, respectively, have been the subject of recent investigative activities. Four potential EAs have previously been identified for the Site (Allendale, Lyman Mill, Manton, and Dyerville). These EAs are located progressively farther south within the reach of the south-flowing Woonasquatucket River constituting the Site. Assapumpset Brook (and Pond) is not part of the Woonasquatucket River itself, but rather a tributary to the river. The Greystone Mill Pond is upstream of the Site on the Woonasquatucket River (an upstream location). Sediment and soil samples were collected just north of the source area (north of Route 44). These sample locations are considered to be part of the upstream (Greystone) data set. Greystone is the farthest upstream of the source area and Dyerville is the most downstream area from the source area. The reach of the Woonasquatucket River within the Allendale EA is both adjacent to and immediately downstream of the source area. The graphical presentation of Site data is intended to provide perspective with respect to the nature and location of likely Site-related impacts. This graphical presentation is not used to screen out or to otherwise adjust the list of analytes that are considered in the risk assessment. The selection of COPCs for the risk assessment is based on all of the available environmental data that are representative of current and future conditions.

2.3.1 Dioxin/Furan, HCX, and Co-planar PCB TEQs

Figures 7 through 17 present histogram summaries of dioxin/furan, HCX, and co-planar PCB toxicity equivalent concentrations (TEQs) in sediment, surface water, floodplain and

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bank soil, and biota tissue including white sucker [brown bullhead instead of white sucker in Assapumpset only], American eel, largemouth bass, crayfish, emerging insects, earthworms, tree swallow eggs, and tree swallow nestlings. The TEQs are media-specific concentrations that are normalized to the toxicity of the 2,3,7,8-TCDD congener, generally considered to be the most toxic of the dioxin, furan, and dioxin-like compounds. The TEQs are calculated by multiplying the medium-specific concentration of each congener or congener group by a Toxicity Equivalence Factor (TEF) and summing those products. The TEF is a measure of the toxicity of a particular congener or congener group relative to toxicity of 2,3,7,8-TCDD. In simple terms, the dioxins/furans TEQ indicates the concentration of 2,3,7,8-TCDD that would have the same toxicity as the mixture of dioxins and furans being evaluated.

The TEFs used in the development of Figures 7 through 17 are the mammalian TEFs for dioxins, furans, and dioxin-like (coplanar) PCBs as published in Van den Berg et al., 1998. The application of the mammalian TEFs in the histograms is intended to provide a consistent means of presenting the nature and extent of contamination. It is not intended to imply that mammalian TEFs are appropriate for estimating risks for ecological receptors and TEFs for birds and fish are also employed as appropriate in evaluating risks. The method used to quantify TEQs for the BERA is described in further detail in Section 3.

Each of the figures presents, for a single medium, average concentrations of each of these three analyte groups for the background and reference areas, Greystone and Assapumpset, respectively, and for potential exposure areas EA1 (Allendale), EA2 (Lyman Mill), EA3 (Manton), and EA4 (Dyerville) in that sequence. In addition, for tree swallow eggs, data are provided from the Fire Station reference area (for 2002 only). Each histogram presents a graphical representation of concentrations at upstream, near-source, and downstream locations (from left to right). All of the data in Figures 7 through 17 are presented in picograms per gram (or parts per trillion (ppt)) so that data may be compared directly across media. All of the concentration data in Figures 7 through 17 are presented on a logarithmic scale – therefore, values that are separated by a major tick-mark on the y-axis have concentrations that differ by a factor of ten, those separated by two tick-marks would differ by a factor of one hundred. EAs where analytical data are not available for a given chemical group are indicated as "Not Analyzed" in the figures.

Several conclusions may be reached from Figures 7 through 17:

- In sediment (Figure 7), dioxin/furan 2,3,7,8-TCDD toxic equivalency (TCDD TEQ) and HCX TCDD TEQs are at least one hundred times higher in Allendale than in the two background/reference areas, and there appears to be a decreasing trend in concentrations moving downstream from Allendale. This suggests a Site-related

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impact on dioxins/furans and HCX in sediments in areas adjacent to and downstream of the source area.

- In sediment (Figure 7), no specific spatial trend for co-planar PCB TCDD TEQs was observed for sediment. This would suggest there is not a Site-related impact on concentrations of co-planar PCBs in sediment in areas adjacent to and downgradient of the source area.

- Dioxin/furan TCDD TEQs in sediment in all areas tested are typically five-hundred to one-thousand times higher than the corresponding HCX TCDD TEQs (Figure 7).

- In surface water (Figure 8), dioxin/furan TCDD TEQs are approximately one hundred times higher in Allendale and approximately ten times higher in Lyman Mill than in the Greystone background area.

- In floodplain and bank soils (Figure 9), dioxin/furan TCDD TEQs are over ten times higher in Allendale and approximately ten times higher in Lyman Mill than in the Greystone background area. No spatial trend for co-planar PCB TCDD TEQs was observed for floodplain soils.

- In white sucker (Figure 10), American eel (Figure 11), and largemouth bass (Figure 12) (no data for Allendale for largemouth bass), dioxin/furan TCDD TEQs are generally ten to one-hundred times higher in Allendale and Lyman Mill than in the two background/reference areas. These average TCDD TEQs are approximately 100 pg/g or 100 ppt. HCX was detected in white sucker and American eel in Allendale and Lyman Mill but not in the background or reference areas. The TCDD TEQs for HCX are equal to or less than 0.01 ppt. Dioxin/furan TCDD TEQs in white sucker, American eel, and largemouth bass (no data for largemouth bass in Allendale) in Allendale and Lyman Mill are generally higher than either the HCX TCDD TEQs or the co-planar PCB TCDD TEQs in those same reaches. The data suggest a Site-related impact on dioxin/furan and HCX concentrations in fish in Allendale and Lyman Mill.

- The average concentrations of co-planar PCB TCDD TEQs in white sucker (Figure 10), American eel (Figure 11), largemouth bass (Figure 12), crayfish (Figure 13), and emerging insects (Figure 14) from Allendale and/or Lyman Mill are higher than the corresponding average concentration in Greystone. These data suggest a possible Site-related impact on co-planar PCB TCDD TEQs in the three fish species, crayfish, and emerging insects from Allendale and/or Lyman Mill.

- In crayfish (Figure 13), average dioxin/furan TCDD TEQs in Allendale and Lyman Mill are approximately thirty to fifty times higher than the corresponding average in

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the Greystone background area. Average concentrations of HCX TCDD TEQs in Allendale and Lyman Mill are approximately five to eight times higher than the corresponding average concentration in the Greystone background area. These data suggest a Site-related impact on dioxins/furans and HCX concentrations in crayfish in Allendale and Lyman Mill.

- In emerging insects (Figure 14), average dioxin/furan TCDD TEQs in Lyman Mill is approximately twenty-five times higher than the corresponding average in the Greystone background area. HCX was detected only in Lyman Mill emerging insects. These data suggest a Site-related impact on dioxins/furans and HCX concentrations in emerging insects in Allendale and Lyman Mill.

- In earthworms (Figure 15), average dioxin/furan TCDD TEQs in Allendale and Lyman Mill are between one and two orders of magnitude higher than the corresponding average in the Greystone background area. HCX was only detected in Allendale and Lyman Mill earthworm tissues; however, the average co-planar PCB TCDD TEQ in Greystone earthworms was approximately five times higher than in Lyman Mill (earthworm samples were not analyzed for PCB congeners in Allendale). These data suggest a Site-related impact on dioxin/furans and HCX concentrations in earthworms in Allendale and Lyman Mill.

- In general, the dioxin/furan TCDD TEQs for tree swallow eggs collected at Allendale, Lyman Mill, and Manton are one order of magnitude greater than those for either Greystone (background) or Fire Station (reference) (Figure 16). The average dioxin/furan TCDD TEQs in Allendale in 2000 was approximately 20 times higher than the corresponding average in Greystone; in 2001, it was approximately 5 times higher than the corresponding background average. In Lyman Mill, the average dioxin/furan TCDD TEQ in 2001 was approximately 10 times higher than the corresponding average in Greystone. In Manton, the average dioxin/furan TCDD TEQ in 2003 was approximately 40 and 8 times higher than the corresponding background and reference averages, respectively. HCX and co-planar PCBs were only detected in tree swallow eggs collected from Allendale and Lyman Mill; these compounds were not analyzed for in Fire Station or Manton samples and were not detected in egg sample collected from Greystone. These data suggest a Site-related impact on dioxins/furans in tree swallow eggs in Allendale, Lyman Mill, and Manton.

- In general, the dioxin/furan TCDD TEQs for tree swallow nestling tissue collected at Allendale, Lyman Mill, and Manton are one order of magnitude greater than those for the upgradient background area in Greystone (Figure 17). The average dioxin/furan TCDD TEQs detected in Allendale in 2000 was approximately 40 times higher than the corresponding average in Greystone. In 2001, the average dioxin/furan TCDD TEQs in Allendale and Lyman Mill were approximately 60 times higher than the corresponding background average. The average dioxin/furan TCDD TEQ for

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Manton in 2003 is approximately 20 times higher than the average background concentration in 2001 (tree swallow nestlings were not sampled from Greystone in 2003). Tree swallow nestling tissue chemistry data from Lyman Mill were only collected in 2001 and data from Manton were only collected in 2003. HCX was not detected in samples from Allendale or Greystone in 2000, but was detected in all three areas sampled (Greystone, Allendale, and Lyman Mill) in 2001. Average HCX TCDD TEQs for Allendale and Lyman Mill were approximately 2 times higher than the corresponding average for Greystone. Average co-planar PCB TCDD TEQs were similar between years from 2000 to 2001 in samples collected from Allendale and Greystone. Average co-planar PCB TCDD TEQs for Allendale and Lyman Mill were approximately 4 to 7 times higher than corresponding average concentrations detected in samples from the background area. HCX and co-planar PCB data were not collected from Manton. These data suggest a Site-related impact on dioxins/furans in tree swallow nestlings in Allendale, Lyman Mill, and Manton, and a Site-related impact on HCX and co-planar PCB concentrations in Allendale and Lyman Mill.

2.3.2 Dioxin/Furan and Co-planar PCB Congener Concentrations

Figures 18 through 28 present selected dioxin and co-planar PCB congener concentrations (raw data, not adjusted to 2,3,7,8-TCDD equivalents) for sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, emerging insects, earthworms, tree swallow eggs, and tree swallow nestlings. The purpose of these figures is to compare and contrast the dioxin/furan "signature" and co-planar PCB "signature" among the background and reference areas and the potential EAs. The "signature" is determined by which congener(s) is present in the highest concentrations and how the average concentrations of specific congeners compare to each other.

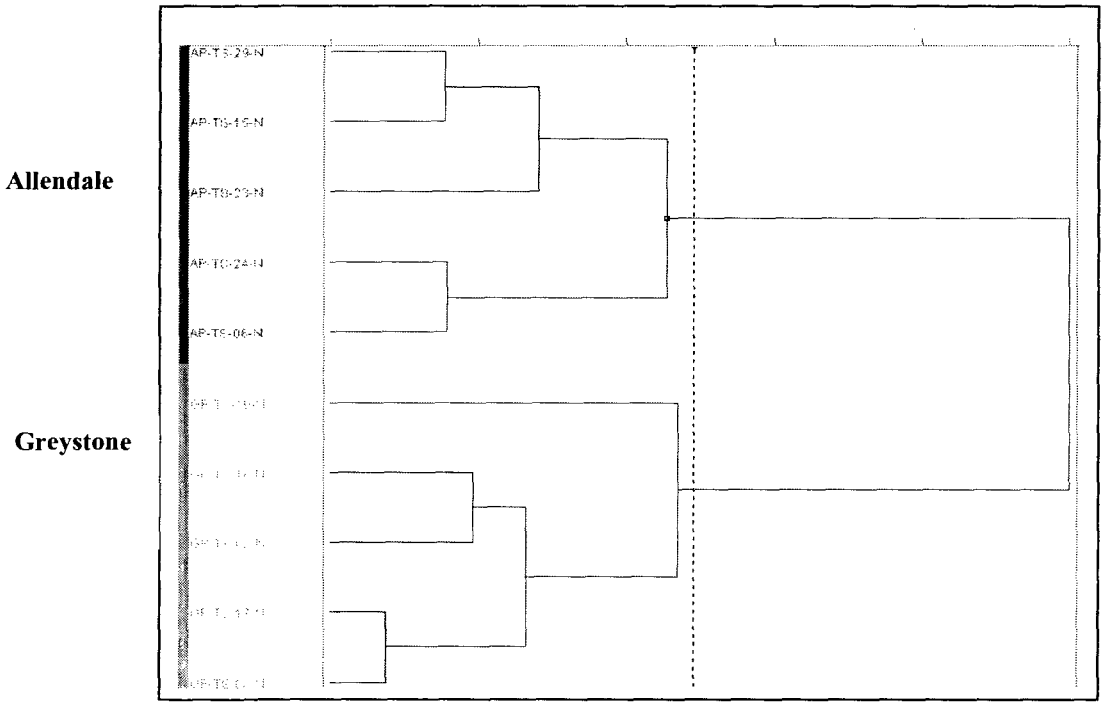
For example, in Figure 18, in the Assapumpset reference area, the four dioxin/furan congeners with the highest sediment concentrations are, (in descending order), octa-chlorinated dibenzodioxin, 1,2,3,4,6,7,8-hepta chlorinated dibenzodioxin, octa-chlorinated dibenzofuran, and 1,2,3,4,6,7,8-hepta chlorinated dibenzofuran. The 2,3,7,8-TCDD congener (i.e., most important from a toxicological perspective) concentration is only about 1% of the concentration of octa-chlorinated dibenzo dioxin, the congener with the highest concentration. In contrast, for Allendale, the 2,3,7,8-TCDD congener has the highest average concentration (approximately 300 times higher than in Assapumpset). Allendale has a similar distribution of the hepta- and octa-chlorinated dibenzodioxins and dibenzofurans as does Assapumpset. The following conclusions may be drawn from Figures 18 through 28.

- There is a pronounced difference in dioxin/furan "signature" between the background and reference areas and Allendale, Lyman Mill, and Manton (for those media with congener data) for sediment (Figure 18), floodplain and bank soil (Figure 20), white

sucker (Figure 21), American eel (Figure 22), largemouth bass (Figure 23), crayfish (Figure 24), emerging insects (Figure 25), and earthworms (Figure 26). In virtually all cases, the average 2,3,7,8-TCDD concentrations are at least 100 times higher in Allendale and Lyman Mill Reaches than in the background and reference locations. Average 2,3,7,8-TCDD concentrations detected in tree swallow eggs and nestlings collected from Allendale, Lyman Mill, and Manton were also elevated relative to the reference and background areas (Figures 27 and 28). Average 2,3,7,8-TCDD concentrations detected in tree swallow eggs collected from Allendale (from 2000 to 2001), Lyman Mill (2001 data only), and Manton (2003 data only) were approximately 3 to 40 times higher than concentrations measured during the same years in the Greystone background area and approximately 3 to 200 times higher than concentrations detected in the Fire Station reference area (Figure 27). The average 2,3,7,8-TCDD concentrations detected in tree swallow nestlings from Allendale in 2000 and 2001 and Lyman Mill in 2001 were approximately 90 to 100 times higher than the concentrations measured in the background area during the same years (Figure 28). The average 2,3,7,8-TCDD concentration measured in Manton tree swallow nestlings in 2003 was approximately 40 times higher than the average background concentration measured in 2001 (background data for nestlings are not available for 2003). These data suggest a Site-related impact on 2,3,7,8-TCDD concentrations for all of the media evaluated for Allendale, Lyman Mill, and Manton.

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- Hierarchical cluster analysis of PCB congeners in tree swallow nestling tissue from demonstrated a pronounced difference in the PCB congener "signature" between the Greystone background area and Allendale in 2000 (Battelle, 2000). As seen in the figure, the distance measures (which were derived based on the specific congeners and their relative magnitudes) was much greater between than within the two EAs. While many dioxin, furan, and PCB congeners were detected in both Greystone and Allendale swallow nestling tissue as approximately similar concentrations (Figure 28)



Hierarchical cluster analysis of PCB congener distributions in nestling samples collected at Greystone and Allendale Ponds.

TCDD was over an order of magnitude higher in concentration in the Allendale EA. This same pattern exists for the 2001 tree swallow nestling tissue data as well (Figure 28).

- In 2000, 2,3,7,8-TCDD was not detected in tree swallow stomach content samples collected from the Greystone background location, but was detected in samples collected from Allendale at 71.4 pg/g (Appendix M, Table 8). Detected concentrations of other dioxin/furan congeners in 2000 in stomach content samples were relatively similar between Greystone and Allendale. In 2001, 2,3,7,8-TCDD concentrations detected in tree swallow stomach content samples were approximately 40 and 20 times higher in Allendale and Lyman Mill, respectively, than in Greystone.

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The average 2,3,7,8-TCDD concentration detected in tree swallow stomach samples collected from Manton (in 2003) was approximately 12 times higher than the corresponding 2001 Greystone concentration. Concentrations of other dioxins/furan congeners detected in 2001 in Greystone, Allendale, and Lyman Mill, and in 2003 in Manton, were relatively similar, with slightly elevated concentrations of most congeners occurring in Allendale relative to the other areas (Appendix M, Table 8).

- It appears that the relative concentrations of the three co-planar PCBs shown on the histograms and the actual magnitude of those concentrations are reasonably consistent among the background/reference areas and the potential EAs for sediment (Figure 18), surface water (Figure 19), floodplain and bank soil (Figure 20), and earthworms (Figure 26). However, the concentrations of PCB-77 in sediment are approximately 10 to 15 times higher in Allendale and Lyman Mill than in Greystone. This suggests possible Site-related impact on co-planar PCB concentrations in sediment. The concentrations of PCB-77 and PCB-126 measured in tree swallow nestling samples collected from the site exposure areas were approximately 4 to 7 times higher than corresponding concentrations measured in the background samples (Figure 28), indicating potential site-related impact. Co-planar PCB concentration data were not collected from background/reference tree swallow eggs (Figure 27).

- Among the biota sampled, tree swallow eggs and nestlings exhibited the highest average concentrations of 2,3,7,8-TCDD.

The downstream migration of contaminated sediments associated with the April 2001 breach of the Allendale Dam likely disrupted existing quasi-equilibrium conditions between biota tissue and bed sediment and increased the bioavailability of bioaccumulating compounds. Consequently, the limited April 2001 fish tissue data were reviewed in order to evaluate whether the June 2001 fish tissue data are consistent with long term conditions in the river. The following analysis is limited to dioxin, furan, and co-planar PCB congeners because only these groups were analyzed for in the April 2001 fish samples.

The single white sucker sample collected from Allendale Pond in April 2001 (AP-CC-06) was small (90 grams) with a lipid content of only 1.8 percent; the ten Allendale Pond white sucker samples collected in June 2001 were considerably larger (900 – 1,450 grams) and their lipid content ranged from 5.2 to 8.7 percent. Table A-16 in the Data Evaluation Report (DER, MACTEC, 2003), presents the analytical chemistry for all Allendale Pond fish tissue samples, including the April 2001 fish. The following observations were derived from a comparison of the analytical tissue data for AP-CC-06 and the two fall fish composite samples (AP-SC-01/-02/-03 and AP-SC-04/-05) with the ten white sucker samples collected in June 2001:

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- HCX was detected in 9 of 10 June 2001 ("post-breach") fish samples and in none of the April 2001 ("pre-breach") samples.
- 2,3,7,8-TCDD was detected in all samples and concentrations between pre- and post-breach samples appear to be similar (0.3 ug/kg in AP-CC-06) ; range in June 2001 white suckers - 0.079 – 0.8 ug/kg.
- TCDD and 1,2,3,4,6,7,8-HpCDD were the only dioxin/furan congeners detected in the pre-breach white sucker sample, 2,3,7,8-TCDF and OCDF were also detected in one or more of the composite fall fish samples; between 1 and 9 dioxin and furan congeners were detected in the post-breach white sucker samples.
- Co-planar PCB congeners were detected in all fish where analyzed (i.e., all pre-breach and 3 of 10 post-breach samples).
- PCB 126, the congener with the highest TEF for fish, was detected at higher concentrations in pre-breach fish – 0.35 ug/kg in AP-CC-06; range in post-breach samples is 0.083 – 0.12 ug/kg; PCB 126 was detected in fall fish composites samples at higher concentrations (e.g., 0.76 and 0.77 ug/kg).

The two white sucker samples collected from Lyman Mill Pond in April 2001 (LP-CC-07 and LP-CC-08) were comparable in weight (1,673 and 1,247 grams, respectively) to those collected in June 2001(range of 900 to 1,800 grams). The lipid contents in pre-breach white sucker fish are 4.8 and 3.5 percent (Table A-19; MACTEC, 2003) the range of lipid content in the ten Allendale Pond white sucker samples collected in June 2001 is 5.36 – 12.1 percent. Table A-19 in the Data Evaluation Report (DER, MACTEC, 2003), presents the analytical chemistry for all Lyman Mill Pond fish tissue samples, including the April 2001 fish. The following observations were derived from a comparison of the analytical tissue data for LP-CC-07 and LP-CC-08 with the ten white sucker samples collected in June 2001:

- Similar to the Allendale Pond results, HCX was detected in 9 of 10 June 2001 post-breach fish samples and in none of the April 2001 pre-breach samples.
- 2,3,7,8-TCDD was detected in all samples and concentrations between pre- and post-breach samples appear to be similar (0.42 and 0.46 ug/kg in LP-CC-07 and LP-CC-08, respectively) ; range in June 2001 white suckers - 0.21 – 1.4 ug/kg.
- TCDD, 1,2,3,4,6,7,8-HpCDD, and TCDF were the only dioxin/furan congeners detected in the pre-breach white sucker sample; between 4 and 12 dioxin and furan congeners were detected in the post-breach white sucker samples.
- Co-planar PCB congeners were detected in all fish where analyzed (i.e., all pre-breach and 3 of 10 post-breach samples).
- PCB 126 was detected at higher concentrations in pre-breach fish – 1.1 and 8.7 ug/kg for LP-CC-07 and LP-CC-08, respectively; range in post-breach samples is 0.09–0.24 ug/kg.

2.3.3 Polycyclic Aromatic Hydrocarbon (PAH) Concentrations

Figures 29 through 33 present the PAH "signature" for the background/reference areas and potential EAs for sediment, floodplain and bank soil, white sucker, American eel, and largemouth bass. Unlike Figures 7 through 28, the concentration scale for these figures is arithmetic or linear, rather than logarithmic because there is less variability (generally less than 10-fold differences rather than 100-fold or 1000-fold differences) among the PAH compounds, and among concentrations at the reference locations and the potential EAs. This makes it possible to present all of the data on a single arithmetic scale.

The following conclusions can be drawn from these PAH histograms:

- In sediment (Figure 29), fluoranthene, pyrene, benzo(b)fluoranthene, and chrysene are consistently present at the highest concentration compounds among the PAHs. Overall, the relative distribution of compounds (signature) looks very similar for the background/reference areas and the potential EAs. However, there is variation in the magnitude of reported concentrations, with Assapumpset and Manton showing the lower concentrations. In general, Greystone, Allendale, and Lyman Mill have similar concentrations of PAHs (i.e., no more than a two-fold difference). There does not appear to be any trend that would suggest a Site-related impact on PAH concentrations in sediment in Allendale or downstream EAs.

- In surface water, PAHs were generally not detected. Therefore, no histogram is presented.

- In floodplain soil and bank soil (Figure 30) (Greystone, Allendale, and Lyman Mill data available), fluoranthene, pyrene, and phenanthrene are consistently the highest concentration compounds among the PAHs. Overall, the relative distribution of compounds (signature) looks very similar for the background /reference area and the potential EAs. One exception is the concentration of acenaphthylene in Lyman Mill, which appears to be approximately ten times higher than in Allendale and Greystone Reaches. As can be seen by comparing the PAH signatures for Lyman Mill sediment (Figure 29) and Lyman Mill floodplain (bank soil not evaluated for that reach) (Figure 30), the PAH signatures are similar and the concentrations are somewhat higher in soils (individual compounds at approximately 12 mg/kg) than in sediment (individual compounds at approximately 2.5 mg/kg).

- In fish (white sucker (Figure 31), American eel (Figure 32), and largemouth bass (Figure 33)), phenanthrene, fluoranthene, and 2-methylnaphthalene are consistently present in the highest concentrations among PAHs. Concentrations of phenanthrene and fluoranthene vary somewhat across fish species (approximately 8 ug/kg to 22 ug/kg phenanthrene in white sucker; not detected to approximately 16 ug/kg

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phenanthrene in American eel; and not detected to approximately 9 ug/kg in largemouth bass). Naphthalene and 2-methylnaphthalene concentrations in all three fish species are higher in Assapumpset than in any other reach. The PAH fingerprint in fish tissue is similar to the sediment fingerprint observed, with the heavier molecular weight PAHs comprising a somewhat smaller portion of the total PAH concentration in fish than in sediment. Concentrations of PAHs in fish tissue are approximately 500 to 1000 times lower than corresponding concentrations in sediment.

- There are no specific trends that would suggest a Site-related impact on PAH distribution or concentrations in surface water, sediment (Figure 29), floodplain and bank soil (Figure 30), white sucker (Figure 31), American eel (Figure 32), or largemouth bass (Figure 33) at or adjacent to the source area within Allendale Reach.

2.3.4 Pesticide/PCB Aroclor Concentrations

Figures 34 through 43 summarize average concentrations of select pesticides (dichlorodiphenyltrichloroethane (DDT), dichlorodiphenyldichloroethane (DDD), dichlorodiphenyldichloroethylene (DDE), dieldrin, chlordane, heptachlor) and PCB Aroclors in sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, earthworms, tree swallow eggs, and tree swallow nestlings. These data are presented with a logarithmic scale for concentration. The following conclusions may be drawn from these histograms.

- In sediment (Figure 34), chlordane (except in Dyerville) and the PCB Aroclors 1254 and 1268 are consistently present in the highest concentrations among pesticides and PCBs. The signature of DDT, DDD, and DDE appears similar in sediment from all background/reference and potential EAs. The average sediment concentration of dieldrin in Allendale appears to be approximately 4-fold higher than in the other reaches. Heptachlor seems to be unique to Allendale and Lyman Mill. In Allendale, Aroclors that were not observed in other reaches are present (Aroclors 1016, 1221, 1232, 1242, and 1248). The average concentrations of Aroclor 1254 in Allendale, Lyman Mill and Dyerville are each greater than the corresponding concentrations in the background and reference areas. This suggests a Site-related impact on Aroclor 1254 concentrations in sediments in Allendale, Lyman Mill, and Dyerville.

- For surface water (Figure 35), as would be expected with the low solubilities of Aroclors and the persistent pesticides, these constituents were typically not detected. However, Aroclor 1254 was only detected in Allendale surface water.

- In floodplain soil and bank soil (Figure 36), the signature and concentrations of DDT, DDD, DDE, dieldrin, chlordane, and heptachlor appear similar in sediment from all background/reference and potential EAs. Aroclors 1254 and 1268 are detected in

Greystone and Lyman Mill (with higher concentrations in Greystone). In Allendale, Aroclors 1242, 1248, and 1254 are detected, all with average concentrations higher than in any of the other reaches.

- In white sucker (Figure 37), American eel (Figure 38), and largemouth bass (Figure 39), Aroclor 1254, chlordane, and Aroclor 1268 are consistently the highest concentration compounds among pesticides and PCBs. Overall, concentrations of Aroclor 1254 and chlordane appear to be higher in Allendale and Lyman Mill than in the background and reference areas. The signature and concentrations of DDT, DDD, DDE, dieldrin, chlordane, and heptachlor appear similar in sediment from the background/reference and potential EAs. Concentrations of these pesticides are consistent in all three fish species. Chlordane concentrations are generally 100 ug/kg or above (greater than 1000 ug/kg in Lyman Mill for white sucker and largemouth bass). Aroclor 1254 concentrations are generally 100 ug/kg or above consistently (greater than 1000 ug/kg for white sucker in Allendale and Lyman Mill and for largemouth bass in Lyman Mill). The data suggest a Site-related impact on Aroclor 1254 concentrations in the three fish species in Allendale and Lyman Mill.

- In crayfish (Figure 40), chlordane and Aroclors 1254 and 1268 are detected consistently and at similar concentrations in Greystone, Allendale, and Lyman Mill (average concentrations appear lower in Assapumpset). Aroclor 1254 average concentrations appear to be highest in Allendale and Lyman Mill. Concentrations of chlordane and Aroclor 1254 in crayfish are roughly ten-fold lower than in fish.

- In earthworms (Figure 41), chlordane and Aroclor 1254 and 1268 are detected consistently and at similar concentrations in Greystone, Allendale, and Lyman Mill. Chlordane and Aroclor 1254 average concentrations appear to be highest in Allendale. Concentrations of chlordane and Aroclor 1254 in earthworms are roughly ten-fold lower than in fish.

- In tree swallow eggs (Figure 42) and tree swallow nestlings (Figure 43), the signature and concentrations of DDT, DDD, DDE, dieldrin, chlordane, Aroclor-1254, and Aroclor-1268 appear similar in Greystone, Allendale, and Lyman Mill.

- The predominant pesticide (chlordane) and Aroclors (1254 and 1268) detected in fish (Figures 37-39), crayfish (Figure 40), and earthworms (Figure 41), are consistent with the predominant pesticide (chlordane) and Aroclors (1254 and 1268) detected in sediments (Figure 34). The concentrations of the chlordane and Aroclors 1254 and 1268 are consistent between sediment and the three fish species.

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2.3.5 Toxic Metals

Figures 44 through 51 summarize average concentrations of select inorganics (arsenic, cadmium, chromium, lead, and mercury) in sediment, surface water, floodplain and bank soil, white sucker, American eel, largemouth bass, crayfish, and earthworms. These metals were identified in the SERE as those most likely to pose a threat to ecological receptors. Overall, there is a small amount of variability in inorganics concentrations among the background and reference areas and EAs for any given medium. Therefore, the inorganics data are presented using an arithmetic scale. The following conclusions may be drawn from these inorganics histograms.

- In sediment, arsenic, cadmium, and mercury concentrations vary very little among the background and reference areas and the four potential EAs (Figure 44). Average concentrations of lead vary among the reaches, with the highest average concentrations reported for Allendale and Lyman Mill. The average concentration of lead in Allendale is approximately double the concentration in Greystone. Chromium concentrations in sediment vary from less than 20 mg/kg in Assapumpset to approximately 150 mg/kg in Dyersville, with no apparent spatial trend. There is little indication of a Site-related impact on inorganics (those presented in the histograms) concentrations in sediments adjacent to and downstream of the source area.

- In surface water, average concentrations of arsenic, cadmium, chromium, and mercury are consistently far below 10 ug/liter in the background and reference areas and in Allendale and Lyman Mill (Figure 45). Average concentrations of lead in surface water are all below 5 ug/liter except in Allendale (approximately 12 ug/l). There is little indication of a Site-related impact on inorganics (those presented in the histograms) concentrations in surface water adjacent to and downstream of the source area.

- In floodplain soil and bank soil, there is no indication that inorganic concentrations in soils adjacent to and downstream of the source area have been impacted by the Site (Figure 46).

- In white sucker, American eel, and largemouth bass, average concentrations of arsenic, cadmium, chromium, lead, and mercury are all below 0.5 mg/kg in all areas tested (Figures 47 - 49). There are no consistent spatial trends that would suggest that fish adjacent to and downstream of the source area have been impacted by a release of the inorganics that are presented in these histograms. For the American Eel, average concentrations of lead in Allendale, Lyman Mill, and Dyersville appear to be somewhat higher than those in the background and reference areas (Figure 48). However, these concentrations are very low (less than 0.5 mg/kg). No Site-related impact is indicated.

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- In crayfish, all average concentrations of inorganics are below 1 mg/kg (Figure 50). There is one apparent spatial trend, with lead concentrations appearing to increase from Greystone to Allendale to Lyman Mill. However, the difference between the average concentration at Greystone and Lyman Mill is only approximately 0.3 mg/kg. Therefore, there is no indication of a substantial Site-related impact on inorganics concentrations in crayfish adjacent to or downstream of the source area.

- In earthworms, there are no spatial trends that indicate that inorganic concentrations in earthworms adjacent to or downstream of the source area have been impacted by the Site (Figure 51).

In summary, dioxins and furans, HCX, and possibly PCB-77 appear to be the primary chemical parameters that are detected in environmental media with frequency of detection and concentrations that are indicative of Site-related impacts.

In addition to the histograms discussed above, Figures 52 through 57 present a "plan-view" representation of the concentration distribution of dioxin equivalents, HCX, total PCBs, total PAHs, total pesticides, and metals in total toxic units (defined and discussed later in the BERA) in sediment and surface water. These figures summarize the nature and extent of contamination for these analytical parameters.

2.3.6 Volatile Organic Compounds

In September 1999, the U.S. Geological Survey conducted vapor diffusion sampling at the Site to identify possible discharge areas for contaminated groundwater (Church et al., 2000). Water-to-vapor diffusion samplers were installed on September 8 and 9, 1999 at 115 locations at the Site. Sixty-two samplers (VS-001 through VS-062) were placed in bottom sediments along a 2,250-foot stretch of the Woonasquatucket River, downstream from the U.S. Route 44 bridge; 43 samplers (VS-063 through VS-077 and VS-088 through VS-115) were installed in sediments in the former mill raceway (located approximately 300 to 400 feet east and parallel to the river), nine samplers (VS-078 through VS-086) were installed in sediments of the cross channel between the river and former mill raceway (approximately feet 1400 south of the U.S. Route 44 bridge); and one sampler (VS-087) was placed in water in the cross channel (above VS-079). One hundred and twelve samples (including eight duplicate samples) were collected on September 21 and 22, 1999 from 104 samplers that were retrieved. The vapor samples were analyzed on site for VOCs (including benzene, chlorobenzene, ethylbenzene, tetrachloroethylene [PCE], 1,1,1-trichloroethane [1,1,1-TCA], trichloroethylene [TCE], toluene, meta/para-xylene, and ortho-xylene).

VOCs were detected in samples collected from 84 of the 104 samplers retrieved. TCE and PCE were the principal VOCs detected. The highest concentrations of TCE and PCE were generally detected in samples collected along the Woonasquatucket River between

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500 and 600 feet downstream of the bridge (west of the northern portion of Cap #1). TCE and PCE were detected at maximum concentrations of 182,000 ppb and 1,390,000 ppb, respectively, in sample VS-013. In the most upstream samples (collected from the Woonasquatucket River between 0 and 400 feet downstream of the bridge), TCE and PCE were detected at maximum concentrations of 25 and 156 ppb, respectively. PCE was not detected in the three most upstream samples (collected between 0 and 150 feet downstream of the bridge), while TCE was detected at a maximum concentration of 25 ppb in these samples. Between approximately 650 and 1,800 feet downstream of the bridge, TCE and PCE were detected at maximum concentrations of 49 and 136 ppb, respectively. In the most downstream samples collected from the Woonasquatucket River (between approximately 1800 and 2,200 feet downstream of the bridge), concentrations of PCE were relatively elevated (maximum of 1,470 ppb), while concentrations of TCE (maximum of 42 ppb) were relatively consistent with the other areas sampled.

Benzene, toluene, meta/para-xylene, and ortho-xylene were each detected in only one sample at concentrations ranging from 9 ppb (by volume) (benzene in VS-107) to 430 ppb (xylene in VS-019). Chlorobenzene was detected in four samples only, at a maximum concentration of 260 ppb (in VS-103).

2.3.7 Nutrients

Total nitrogen and total phosphorus concentrations detected in surface water samples collected from the Woonasquatucket River (RWR-SW-5001, RWR-SW-5002, and WRC-SW-4001 through WRC-SW-4004) were compared to natural background concentrations of these nutrients in rivers and streams of the Eastern Coastal Plain, as determined by Smith et al., 2003 using empirical models. Total nitrogen concentrations in surface water samples RWR-SW-5001, RWR-SW-5002, and WRC-SW-4001 through WRC-SW-4004 were approximately 2.5 times higher than the reported natural surface water background concentration of approximately 0.55 mg/L, and total phosphorus concentrations in the samples were approximately 4 to 6 times higher than the respective natural background concentration of 0.015 mg/L. Concentrations of both total nitrogen and total phosphorus were relatively consistent in all samples; there were no observed concentration gradients despite the spatial distribution of the samples (i.e., samples RWR-SW-5001 and RWR-SW-5002, located south of the Greystone Mill Dam and north of the U.S. Route 44 bridge did not have higher concentrations of total nitrogen and total phosphorus than the other samples, which were collected farther downstream, south of the bridge.).

2.4 CONTAMINANTS OF POTENTIAL CONCERN SELECTION

This section describes the process of selecting COPCs for aquatic and terrestrial invertebrates, fish, and wildlife (including birds and mammals). Dioxin and furan congeners, certain PCB congeners, and HCX are most clearly associated with the

Centredale Manor source area and due to the complexity of the risk analysis of ecological effects associated with these compounds; no attempt was made to eliminate any specific congener at the COPC selection stage.

Consistent with USEPA Region I policy, all analytes selected as COPCs in environmental media (based on detection frequency and screening benchmark exceedances criteria) for Site exposure areas were retained in order to estimate total Site risks. (Compounds of Interest (COIs) were also identified for background and reference areas.) Based on the evaluation of trends in the distribution of chemicals in various environmental media (as discussed in Section 1.7), the primary COPCs for site environmental media are dioxin (primarily 2,3,7,8-TCDD) and furan congeners, co-planar PCB congeners (notably PCB 77), PCB mixtures (notably Aroclor 1254), and several pesticides (including chlordane). These specific COPCs are the focus of subsequent discussions in the BERA. Due to differences in the selection criteria and environmental media evaluated, the COPCs selected in the BERA are not identical to those retained in the Baseline Human Health Risk Assessment (BHHRA)

2.4.1 Screening Benchmarks

Protective benchmark concentrations were developed for invertebrate, fish and wildlife receptors to select COPCs. Toxicity benchmarks and Toxicity Reference Values (TRV) values were identified for VOCs, PAHs, non-PAH SVOCs, pesticides, Total PCBs, polychlorinated dibenzodioxins (PCDD)/polychlorinated dibenzofurans (PCDFs), and inorganic chemicals retained as COPCs for the BERA.

In general, toxicity information available in the ECOTOXicological database (ECOTOX) (USEPA, 2003) and other online resources (including Integrated Risk Information System (IRIS) and International Chemical Database (INCHEM)) were reviewed to identify toxicity values for soil invertebrates, benthic invertebrates, plants, fish, mammals, and birds. Toxicity values were identified for both pelagic and demersal aquatic communities, soil communities, mammals, and birds. This information was supplemented with data reported in toxicity benchmark reports prepared for the Oak Ridge National Laboratory (Efroymson et al., 1997; Sample, et al., 1966; Will and Suter, 1995a and 1995b) and other ecotoxicity review documents. Toxicity values were not identified for amphibians and reptiles due to the lack of specific toxicological data for these receptors.

The evaluation of literature toxicity values and identification of benchmarks and TRVs for measurement endpoint receptors considered: (1) ecological relevance of the study, (2) exposure duration, and (3) study endpoints. The evaluation of literature toxicity values for deriving TRVs focused on toxicological data characterizing adverse effects on ecologically relevant endpoints, such as survival, growth, and reproduction. In general,

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benchmark values were identified for surface water, sediment, soil, invertebrates, and fish tissue.

2.4.1.1 Surface Water

The surface water benchmarks used for the screening-level ERA are presented in Appendix D, Table D-1. This table provides a compilation of protective criteria and guidelines from different regulatory agencies that were derived to be protective of aquatic receptors including aquatic macroinvertebrates exposed to contaminants in surface water. The criteria and guidelines are sufficiently conservative that no adjustments to derive no observed adverse effect levels (NOAELs) are necessary. Primary criteria and guidelines used to select surface water benchmarks include: chronic Ambient Water Quality Criteria (AWQC) for freshwater (USEPA, 2002; 1991), RIDEM National Water Quality Criterion (NWQC) for freshwater (RIDEM, 2002), and chronic ecotoxicity thresholds (ETs) for freshwater (USEPA, 1996a). These criteria were used as NOAEL-equivalent toxicity effects endpoints at both the individual and community levels for pelagic (i.e., water column) species.

A 10-fold factor was applied to acute AWQC when chronic AWQC are not available. USEPA (1991) has also identified Lowest Observed Effects Concentrations (LOECs) for many chemicals (including chlorinated solvents) for which there were insufficient data to generate an AWQC; these values are generally accepted as screening surrogates for AWQC. The AWQC and ETs for cadmium, copper, lead, nickel, and zinc were adjusted to a hardness of 25 mg/L as $CaCO_3$ due to the low hardness of the Woonasquatucket River surface water (average hardness for Allendale and Lyman Mill surface water is 28.6 and 29.2 mg/L as $CaCO_3$, respectively).

If no AWQC, ETs, or NWQSs were available for a particular analyte, other sources of information were consulted to derive an appropriate benchmark value. These included the Aquatic Information Retrieval (AQUIRE) database, the Oak Ridge National Laboratories (ORNL) document for screening COPCs for aquatic life (Suter and Tsao, 1996), and the USEPA Ecological Structure Activity Relationships (ECOSAR) software program. Data selected from AQUIRE are based on studies with applicable toxicity endpoints (i.e., survival, growth, reproduction, biomass, and population endpoints). Values from the ORNL document were derived for the purpose of conducting screening-level ERAs and selecting COPCs for further consideration in BERAs. The ECOSAR program estimates effects data for compounds by using known chemical and physical properties of that compound and toxicity data for a related chemical group of compounds.

Surface water benchmarks were selected based on the sequence presented above, with preference given to AWQC or the RIDEM NWQCs (see Table D-1).

2.4.1.2 Groundwater

The TRVs identified for the surface water medium were also used to identify COPCs for overburden groundwater.

2.4.1.3 Sediment

The benchmarks used are presented in Appendix D, Table D-2. This table provides a compilation of criteria and guidelines and is considered to be protective of aquatic receptors exposed to contaminants in sediment. The criteria and guidelines are sufficiently conservative that no adjustment to derive NOAELs is necessary.

Sediment benchmarks for aquatic life were obtained from several sources of information, including: the USEPA sediment quality criteria (SQC) and guidelines (USEPA, 1993a, b, c, d, e; 1988), the Ontario Ministry of the Environment (OME) Lowest Effect Levels (LELs) (Persaud et al., 1996), and the National Oceanic and Atmospheric Administration (NOAA) Effects Range-Low (ER-L) values (Long et al., 1995, MacDonald et al., 2000). Sediment benchmarks protective of aquatic organisms were selected as the lowest value among these if available (see Table D-2).

SQC have been proposed or established by USEPA for a limited number of organic chemicals (USEPA, 1993a, b, c, and d; USEPA, 1988). The USEPA SQC were derived assuming a total organic carbon (TOC) concentration of 10,000 mg/kg (i.e., 1 %), which is conservative as the measured average TOC in exposure area sediments range from 2-10 percent. Since the AWQC are designed to be protective of 95 percent of aquatic species, these sediment concentrations (generated using an equilibrium partitioning approach) are expected to be equally conservative in evaluating adverse effects to benthic organisms (USEPA, 1992b).

The OME LELs represent chemical concentrations below which biological effects are improbable (following an Apparent Effects Threshold approach [USEPA, 1992b]). The OME values were derived largely using data from the Great Lakes and other freshwater bodies. NOAA ER-Ls represent concentrations that may rarely (i.e., 10th percentile) be associated with toxicity to benthic organisms (by the National Status and Trends Program Approach [USEPA, 1992b]). These sediment guidelines are based on a database of primarily marine and estuarine studies. However, in lieu of USEPA SQC or OME LELs, they are generally recognized as applicable for evaluating effects for freshwater aquatic life. These guidelines, which are the model for many ORNL documents that select 10^{th} percentiles for the distribution of effect concentrations, are considered to be equivalent to NOAELs.

If USEPA SQC, OME LELs, and NOAA ER-Ls were not available for a particular analyte, other sources of information were consulted including the ORNL secondary

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chronic values for screening COPCs for effects on sediment-associated biota (Jones et al., 1997).

2.4.1.4 Floodplain Soil

Measures of effect to the floodplain soil invertebrate community associated with the Woonasquatucket River floodplain soils were based on literature-derived toxicological benchmarks. Table D-3 (Appendix D) summarizes the toxicity data available for earthworms and soil invertebrates that were used to select COPCs. The effects primarily considered for earthworms were measures of reproduction (i.e., cocoon formation) or mortality. Reproductive effects data are generally more sensitive than other endpoints; therefore, these data were selected when available. A factor of 0.1 was applied to toxicity data for reproductive effects to derive a NOAEL equivalent. Mortality data are generally based on LC_{50}s, which is the concentration lethal to 50% of a test population. When LC_{50} data were used, a factor of 0.2 was applied to the LC_{50} to derive a Lowest Observed Adverse Effect Level (LOAEL). Another factor of 0.1 was applied to the LOAEL to derive a NOAEL.

Data for soil and litter invertebrates are presented in Efroymson et al. (1997) who derived the 10th percentile of the distribution of various toxic effects thresholds for these receptors. These values were selected preferentially if available; otherwise soil benchmarks summarized by USEPA Region IV were used. For chemicals that lacked empirically-derived benchmarks, soil benchmarks were estimated using Quantitative Structure Activity Relationship (QSAR) or based on uptake models (Table D-3).

2.4.1.5 Biota Tissue

No attempt was made to screen contaminants detected in biological tissue due to the complexity of the potential exposures to wildlife predators. All analytes detected in biological tissue were retained in analysis. Inorganic analytes analyzed for, but not detected in prey tissue, were not retained as sediment or floodplain soil COPCs for wildlife receptors, because it was assumed that they were not bioavailable in the abiotic medium.

2.4.2 Contaminants of Potential Concern for Surface Water

Table 1 summarizes the COPCs selected for surface water; COPCs include several organochlorine pesticides, inorganics, ammonia, and TCDD TEQ for dioxin and furan congeners.

2.4.3 Contaminants of Potential Concern for Groundwater

Table 2 summarizes the COPCs selected for Allendale overburden groundwater. Selected COPCs include tetrachloroethene, trichloroethene, 2,4,6-trichlorophenol, 2,4-

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dichlorophenol, and four PAHs (benzo(k)fluoranthene, chrysene, naphthalene, and pyrene). These COPCs were identified for aquatic invertebrates and fish receptors in the Allendale EA only.

2.4.4 Contaminants of Potential Concern for Sediment

Selected COPCs for sediment are identified in Table 3. Retained analytes include SVOCs, pesticides, PCB Aroclors, inorganics, and TCDD TEQs for dioxins/furans, and PCB congeners. These COPCs were identified for aquatic invertebrates, fish, and piscivorous and omnivorous wildlife receptors. Only the TCDD TEQs for dioxins/furans and PCB congeners were retained as COPCs for insectivorous aquatic wildlife because this class of receptors is only indirectly exposed to sediment contaminants through consumption of emerging aquatic prey.

2.4.5 Contaminants of Potential Concern for Crayfish Tissue

Table 4 identifies the COPCs selected for crayfish tissue. Pesticides, PCB Aroclors, inorganics and TCDD TEQs for dioxins/furans and PCB congeners detected in crayfish tissue were retained as COPCs for aquatic invertebrates and omnivorous and piscivorous wildlife receptors.

2.4.6 Contaminants of Potential Concern for Emerging Insect Tissue

Table 5 summarizes the COPCs selected for emerging insect tissue, which include TCDD TEQ for dioxin/ furan and PCB congeners. These COPCs were identified for aquatic invertebrates and insectivorous wildlife receptors only.

2.4.7 Contaminants of Potential Concern for Tree Swallow Stomach Contents

Table 6 summarizes the COPCs selected for analytes detected in tree swallow stomach content samples; COPCs include TCDD TEQs for dioxin and furan congeners. Similar to the emerging insect medium, these COPCs were identified for aquatic invertebrates and insectivorous wildlife receptors only.

2.4.8 Contaminants of Potential Concern for Floodplain Soil

COPCs retained for floodplain soil are presented in Table 7 and include several phthalates and PAHs, pesticides and PCB Aroclors, inorganics, and TCDD TEQs for dioxin/furan and PCB congeners. These COPCs were identified for floodplain soil invertebrates and vermivorous and omnivorous wildlife receptors.

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2.4.9 Contaminants of Potential Concern for Earthworm Tissue

Table 8 summarizes the COPCs selected for analytes detected in earthworm tissue. Analytes retained as COPCs include pesticides, inorganics, and TCDD TEQs for dioxin/furan and PCB congeners. These COPCs were identified for floodplain soil invertebrates and vermivorous and omnivorous wildlife receptors.

2.4.10 Contaminants of Potential Concern for White Sucker Tissue

Table 9 summarizes the COPCs selected for analytes detected in white sucker tissue. Analytes retained as COPCs include SVOCs, pesticides, inorganics, and TCDD TEQs for dioxin/furan and PCB congeners. These COPCs were identified for demersal fish, and piscivorous and omnivorous wildlife receptors.

2.4.11 Contaminants of Potential Concern for American Eel Tissue

Table 10 summarizes the COPCs selected for analytes detected in eel tissue. Analytes retained as COPCs include SVOCs, pesticides, inorganics, and TCDD TEQs for dioxin/furan and PCB congeners. These COPCs were identified for demersal fish, and piscivorous and omnivorous wildlife receptors.

2.4.12 Contaminants of Potential Concern for Largemouth Bass Tissue

Table 11 summarizes the COPCs selected for analytes detected in largemouth bass tissue. Analytes retained as COPCs include SVOCs, pesticides, inorganics, and TCDD TEQs for dioxin/furan and PCB congeners. These COPCs were identified for piscivorous/pelagic fish, and piscivorous and omnivorous wildlife receptors.

2.4.13 Contaminants of Potential Concern for Tree Swallow Egg Tissue

Pesticides, PCB Aroclors, and TCDD TEQs for dioxin/furan and PCB congeners were selected as COPCs for tree swallow egg tissue (Table 12). These COPCs were identified for insectivorous bird receptors only.

2.4.14 Contaminants of Potential Concern for Tree Swallow Nestling Tissue

Pesticides, PCB Aroclors, and TCDD TEQs for dioxin/furan and PCB congeners were selected as COPCs for tree swallow nestling tissue (Table 13). These COPCs were identified for insectivorous bird receptors only.

2.5 ECOTOXICITY OF CONTAMINANTS OF POTENTIAL CONCERN

This section discusses the general ecotoxicological concerns associated with each class of COPC. The discussion is organized by the broad taxonomic categories of assessment endpoint receptors (Section 2.6), including invertebrates, fish, birds, and mammals.

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2.5.1 Dioxins/Furans

The class of polychlorinated dibenzodioxin compounds consist of 75 isomers that differ in the number and position of the attached chlorine atoms; 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) is the most toxic of the chlorinated dioxin isomers.

2.5.1.1 Mode of Action

Five categories of toxicological effects have been identified for dioxin and dioxin-like compounds (including co-planar PCB congeners) that are of significance from an ecological risk perspective. These effects have all been demonstrated for mammal species and are believed to occur in most other vertebrate groups, as well. These effects are summarized below:

Cytochrome P450 Induction. The ability of 2,3,7,8-TCDD and other compounds with dioxin-like activity to cause the characteristic toxicological effects associated with exposure is believed to be due to the ability of this class of compounds to bind to the aryl hydrocarbon (Ah) receptor (a specific protein in the cytoplasm of body cells). The dioxin/Ah receptor complex has been shown to bind to nuclear DNA, where promotion of gene activity occurs. This results in increased production of a number of proteins, such as cytochrome P450 1A1, which in turn initiates the increased production of P450-dependent enzymes capable of oxidizing both foreign and endogenous substances. Although, the function of this response may be to protect the cell from potentially damaging alien substances, it can affect the metabolism of useful substances like steroid hormones, resulting in disturbances of critical biological functions (Eisler and Belisle, 1996).

The amplified enzymatic activity that is characteristic of Ah-mediated response following dioxin exposure has been used to develop a chemical biomarker of exposure. The addition of ethoxyresorufin to dioxin-exposed cell preparations results in the production of resorufin, a highly fluorescent compound. The fluorescent property of this compound can be used to measure the ethoxyresorufin-O-deethylase (EROD) activity of the preparation. The EROD biomarker can be used to identify historical exposure to dioxin-like compounds in biological samples and has also been used to estimate the relative potency of individual dioxin-like compounds or chemical mixtures.

Immune System Suppression. Dioxins are widely held to have effects on the immune systems of exposed animals. For example, dietary exposure to dioxin, furan and co-planar PCB congeners, measured as TEQs has been correlated with immunosuppression in field experiments with harbor seals (DeSwart et al., 1995; Ross et al., 1996). This type of effect may have contributed to the mass mortalities of seals and dolphins in European waters in the late 1980s and 1990s. The exact mechanism for immune system effects is

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not well understood but dioxin and furan congeners are known to cause hypertrophy of the thymus, an important part of the immune system.

Porphyria. Hepatic porphyria is a condition in which there is a disruption of the process by which the liver produces a compound of the blood pigment hemoglobin. Dioxins and other organochlorine compounds (including pesticides) are known to disrupt the process leading to sensory disorders, paralysis, and psychological effects. Because the porphyrins build up in tissues and feces they are used as biomarkers for dioxin exposure; Fox et al., (1988) detected accumulation of porphyrins in Great Lakes herring gulls that were correlated with high levels of dioxins, furans, and DDE.

Disruption of Vitamin A Metabolism. Dioxins can inhibit the process by which Vitamin A is stored in the liver. Decreased Vitamin A storage, and increased levels in the blood, can result in fetal damage, growth disorders and sterility.

Sex Hormone Effects. A number of persistent organic pollutants have been found to have significant effects on the sex hormones estrogen and testosterone. TCDD decreases the number of estrogen receptors in certain organs in female rats (De Vito et al., 1992) possibly resulting in decreased fertility and increased incidence of tumors in these organs (Kociba et al., 1978). In male rats, TCDD reduces testosterone levels by preventing production of enzymes responsible for increasing testosterone synthesis when levels of the hormone get low (Bookstaff et al., 1990).

2.5.1.2 Ecotoxicology of Specific Taxonomic Groups

Plants. No reports of dioxin toxicity in plants have been identified. Some species of aquatic plants, e.g., the alga (*Oedogonium cardiacum*) and higher plants such as duckweed (*Lemna trisulca*) and *Potamogeton berchtoldii fieber*, have been observed to concentrate dioxins from their surroundings, but did not show any toxic effects (Yockim et al., 1978; Corbet et al., 1988).

Invertebrates. Relatively few experiments have investigated the toxicity of dioxins to invertebrates. Where work has been conducted, the results generally indicate no susceptibility to dioxins (e.g., Yockim et al., 1978; Pruell et al., 1993; Loonen et al., 1996; West et al., 1997). In one case, even where field evidence showed a correlation between exposure to dioxins and mortality in sediment-dwelling amphipods, laboratory experiments with spiked sediments found no effects and suggested that some other factor must have caused the field mortalities (Barber et al., 1998).

Experiments which have shown toxic effects of dioxins in invertebrates include early work on snails and oligochaete worms (Miller et al., 1973), which showed reduced reproductive success when exposed to water spiked to 200 ng/l 2,3,7,8-TCDD. Rhodes et al. (1997) found that soft shelled clams were able to accumulate TCDD in the gonad after

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acute exposure to TCDD in water and noted possible alteration in expression of genes associated increased cell cycling. One study reported acute toxicity in crayfish exposed to TCDD (lethal dose 50 (LD_{50}) of 30-100 ug/kg body weight) and demonstrated cytochrome P450 induction (Ashley et al., 1996). This was the only evidence found during this review for the presence of an Ah-receptor like response to dioxins in invertebrates comparable to that found in vertebrates. Two species of earthworms exhibited no adverse effects when exposed to soil containing 5 mg/kg TCDD for 85 days, however test organisms died at 10 mg/kg (Eisler, 1985). Hahn et al., (1992) were not able to find a functional Ah-receptor in nine invertebrate species and this is a possible reason for the apparent lack of susceptibility to dioxins reported in most invertebrate studies.

Fish. A range of symptoms is shown by fish exposed to dioxin contamination. In the developing embryo-larval stage, TCDD causes an overt toxicity syndrome characterized by edema, hemorrhages, and arrested growth and development culminating in death (e.g., Wisk and Cooper, 1990; Wannamacher et al., 1992). This syndrome is essential similar to blue sac disease. Behavioral responses include reduced feeding, lethargy, unresponsiveness and "head-up" swimming (Merhle et al., 1988).

In general, dioxins are of greatest toxicity to fish embryos, adult life stages for fish exhibit lower sensitivity, with the LD_{50} of TCDD in rainbow trout sac fry being 35 times lower than that in juvenile rainbow trout (Walker and Peterson, 1991). Toxicity in fish tends to be higher for congeners containing four, five, or six chlorine atoms (Wisk and Cooper, 1990), and one study found that octachlorinated congeners had no apparent toxic effects (Berends et al., 1996). It appears that congeners with fewer chlorine atoms tend to be more rapidly metabolized and eliminated (e.g., Muir et al., 1985; Opperhuizen and Sijm, 1990), while more highly chlorinated forms have limited membrane permeability or bioavailability (Gobas and Schrap, 1990). As with higher mammal studies, it appears that 2,3,7,8-TCDD is the most toxic congener to juvenile and adult fish, with toxic effects noted at concentrations as low as 300 ng/kg (whole body) (van der Weiden et al., 1992). Fish eggs also show great sensitivity with LOAEL values as low as 50 ng/kg and LD_{50} values as low as 58 ng/kg based on subsequent effects on hatched fry. This degree of sensitivity is significant because it relates to redistribution of dioxins from maternal tissues to the developing egg cells, which represents the most important route to exposure for ELSs (USEPA, 1993).

Available literature concerning the effects of 2,3,7,8-TCDD on ELS freshwater fish was reviewed. Species examined in the literature included trout (*Salvelinus namaycush*, *S. fontinalis*, and *Oncorhynchus mykiss*), fathead minnow (*Pimephales promelas*), channel catfish (*Ictalurus punctatus*), lake herring (*Coregonus artedii*), Medaka (*Oryzias latipes*), white sucker (*Catastomus commersoni*), northern pike (*Esox lucius*), and zebrafish (*Danio rerio*). Observed 2,3,7,8-TCDD toxicity effects included mortality, edema, hemorrhaging, arrested development, and craniofacial malformations (Walker and

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Peterson, 1994; Walker et al., 1994; Elonen et al., 1998; Walker et al., 1996). Of the fish examined, trout were the most sensitive to 2,3,7,8-TCDD, having $LC_{egg}50$ concentrations ranging from 69 to 179 pg/g egg (Walker et al., 1994; Walker and Peterson, 1994; Walker et al., 1996). Other relatively sensitive species included fathead minnow, channel catfish, and lake herring, with $LC_{egg}50$ concentrations of 539, 644, and 902 pg/g egg, respectively (Elonen et al., 1998). Zebrafish (*Danio rerio*) and northern pike (*Esox lucius*) were least sensitive to TCDD, with $LC_{egg}50$ concentrations of 2,610 and 2,460, respectively (Elonen et al., 1998).

Birds. Bird embryos are considerably more sensitive to TCDD toxicity than are adults (e.g., the LD_{50} of TCDD in chicken embryos is between 100 to 200 times less than the TCDD dose that causes mortality in adults (Peterson et al., 1993). The toxicity of TCDD and other dioxin-like compounds in bird embryos varies widely among different bird species with the chicken being by far the most sensitive of those species tested to date (Brunstrom and Lund, 1988; Nosek et al., 1992, 1993).

One of the earliest links made between dioxin toxicity and observed damage to wildlife populations was around the Great Lakes in Canada and the United States. Increasing numbers of chick deformities were noted amongst cormorants, terns and other fish-eating species both in surviving chicks and embryos that did not hatch. This suite of conditions came to be known as GLEMEDs (Great Lakes embryo mortality, edema and deformity syndrome). The deformities were determined to be very similar to those induced in offspring of hens exposed to dioxins and furans in their feed (Gilbertson et al., 1991) and this observation was linked to concerns about the emissions of dioxins and PCBs from industrial sources such as pulp bleaching processes. Subsequent studies showed correlation between the TCDD-TEQs and effects such as reduced egg hatching, embryotoxicity, deformities (notably crossed bills) and impaired parental behavior (e.g., Tillit et al., 1993; Yamashita et al., 1993; Ludwig et al., 1996).

Laboratory studies have shown other bird species to be susceptible to exposure to dioxins/furans. Chickens and pheasants display decreased egg production, embryotoxicity and cardiovascular malformations (Cheung et al., 1981; Nosek et al., 1992, 1993). Recent work has shown that *in ovo* exposure to dioxins can lead to grossly asymmetric development in the avian brain; a phenomenon seen in field collected herons, cormorants and eagles as well as laboratory hens (Henshel et al., 1997; Henshel, 1998). A least one study has suggested that chickens may be more sensitive to these types of effects on exposure to dioxins and PCBs than wild birds such as cormorants (Powell et al., 1997).

Mammals. Most of the adverse effects attributable to dioxin were first identified from studies of laboratory animals and these initial results appear applicable to wild populations of mammals in field observations and experiments. For example, mink fed on carp from contaminated water in Saginaw Bay, Michigan exhibited clinical signs

associated with dioxin toxicity such as listlessness, anorexia, lowered red blood cell counts, enlarged spleens, livers, and lungs (Heaton et al., 1995). In a review of the literature, Peterson et al., (1993) concluded that decreases in spermatogenesis and the ability to carry a pregnancy to full term are the most sensitive signs of reproductive toxicity in male and female mammals, respectively. Symptoms of developmental toxicity expressed in mammals include decreased growth, structural malformations, functional alterations, and prenatal mortality (Peterson et al., 1993).

2.5.1.3 Toxic Equivalence Factors (TEFs)

It is known that the toxicity of dioxin-like compounds is not equal as defined by their ability to cause specific toxic effects in animals and fish. As a result, the potential toxicity of dioxin mixtures varies depending on the proportions of the individual congeners in the sample. In addition, the relative proportions of each congener will change over time with transport from one environmental compartment to another as a result of differential degradation, metabolism, uptake, or elimination rates. The difficulty in assessing the overall toxicity of dioxin mixtures has been overcome by expressing the toxicity of each congener as a Toxic Equivalency Factor (TEF). The TEF represents the toxicity of the congener relative to 2,3,7,8-TCDD, which is recognized as the most potent dioxin compound. A Toxic Equivalent (TEQ) concentration can be derived by summing the products of the individual congener concentrations and their corresponding TEFs for a given environmental sample. The TEQ values presented in the BERA are reported separately for the sum of dioxin and furan congeners and the sum of PCB congeners.

Several sets of TEFs have been developed based primarily on mammalian laboratory bioassays (Ahlborg et al., 1994; Safe, 1987). The TEF concept has recently been reviewed by the European Centre of Environmental Health of the World Health Organization (ECEH-WHO) and the International Programme on Chemical Safety (IPCS). The consensus-based TEFs for mammals, birds, and fish (Van den Berg et al., 1998) used in this BERA are presented in Table 14.

No consensus-based TEF has been developed for HCX and there is limited information on the relative potency of this compound relative to TCDD. DeCaprio et al., (1987) studied the acute toxicity and *in vitro* activity of HCX in the guinea pig and concluded that HCX is at least 200,000 times less active than TCDD in causing standard pathological effects (including survival, body weight loss, and organ weight changes) in the laboratory. An *in vitro* bioassay study (Viswanathan et al., 1987) demonstrated that HCX was approximately one million times less active than TCDD in causing reversible inhibition of cell growth in a mouse epithelial cell line. Hahn (2001) conducted a biochemical competition assay to assess the relative affinity of HCX relative to TCDD for binding to both fish (trout) and mammal (human) Ah-receptors. Only two concentrations were tested, but based on a qualitative assessment of the data, the relative affinity of HCX in these tests was estimated to be on the order of about 5000 times less

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than TCDD. The preliminary results, which are presented in Appendix H, suggested that HCX competed more effectively with TCDD in the trout Ah-receptor test than with the human Ah-receptor. This study was not designed to evaluate the relative toxicity of HCX or to determine whether the compound is a TCDD agonist or antagonist. Based on the Hahn (2001) study, an interim TEF of 0.0002 was selected for use in the BERA for mammals, birds, and fish (Table 14).

2.5.2 Polychlorinated Biphenyls

PCBs are a class of 209 different congener compounds, each consisting of a chlorinated biphenyl moiety with between one and ten attached chlorines atoms. The chemical and physical properties of the individual congeners are largely determined by the number of chlorines and the position of the substituted chlorines. As the degree of chlorine substitution increases, the aqueous solubility and vapor pressure of the specific congeners decrease.

The fate and transport of PCBs in environmental systems is controlled by distribution or partitioning of PCBs between sediment, suspended particulates, surface water and biota. The observed partitioning of nonionic organic chemicals, such as PCBs, is due to sorption to organic phases, including porewater dissolved organic carbon and sedimentary organic matter. The extent to which congeners are associated with organic matter relative to their dissolved aqueous concentrations is related to their levels of chlorination. The more chlorinated congeners have stronger tendencies to be associated with particulate and dissolved organic matter than the less chlorinated congeners.

The absence of chlorine substitution in the *ortho* (i.e., positions adjacent to the biphenyl bond) position in 20 PCB congeners facilitates the assumption of a planar configuration that is stereochemically similar to the 2,3,7,8-TCDD molecule (Eisler and Belisle, 1996). However, some congeners with 1 or 2 *ortho* chlorines can also assume a planar configuration. There are two categories of coplanar PCB congeners with established consensus-based TEFs: non-*ortho* planar PCBs (including PCBs 77, 81, 126, 169) and mono-*ortho* planar PCBs (including PCBs 105, 114, 118, 123, 156, 157, 169, and 189) (Van den Berg et al., 1998). Collectively, these congeners are referred to as coplanar PCBs in the BERA.

2.5.2.1 Mode of Action

The mode of toxicity of planar and mono-*ortho* PCB congeners is associated with their ability to interact with the Ah receptor (AhR) protein in a manner similar to 2,3,7,8-TCDD. However, some PCB-induced toxicity (including neurological and immunological effects, tissue damage) is not Ah-receptor mediated and a joint role of Ah-receptor dependent and independent mechanisms is believed to be associated with

liver hypertrophy, reproductive effects attributable to steroid hormone imbalance, thyroid hormone imbalance, and immunological effects (ATSDR, 2000).

2.5.2.2 Ecotoxicology of Specific Taxonomic Groups

Invertebrates. PCBs have a wide variety of effects on aquatic organisms. There are significant interspecies differences in sensitivities to PCBs, even among species that are closely related taxonomically (Eisler, 1986a). Most studies of the effects of PCBs on benthic invertebrates have shown effects on mortality, growth, and reproductive impairment.

Fish. Effects of PCBs on fish include mortality, growth-related effects, behavior responses, biochemical alterations, and adverse reproductive effects. Of particular concern are the effects of dioxin-like PCB congeners, which have the same toxic mechanism as 2,3,7,8-TECC (Walker and Peterson, 1991; Zabel et al., 1995a). Similar to TCDD, dioxin-like PCB congeners cause early life-stage mortality associated with blue-sac disease, which involves subcutaneous yolk sac edema (Wisk and Cooper, 1990; Walker et al., 1991).

In addition, numerous field studies have reported increased mortality, pathological anomalies, and biochemical changes in feral fish collected from water bodies contaminated with PCBs (Niimi, 1996). These observations include reduced hatchability and poor survival of larvae taken from feral organisms and reared in the laboratory (Ankley et al., 1991).

Birds. PCBs 77 and 126 are among the most toxic congeners to avian species, with the domestic chicken particularly sensitive to dioxin-like effects associated with these congeners (Eisler and Belisle, 1996).

A substantial amount of research has demonstrated that adverse reproductive effects occur in piscivorous bird populations exposed to PCBs and dioxins in the Great Lakes Region (Jones et al., 1993; 1994; Tillitt et al., 1993; Giesy et al., 1994). The majority of the research has focused on double-crested cormorants because deformities were first observed in this species. Some work has recently been done to evaluate reproductive effects of PCBs in the great blue heron (Sanderson et al., 1994, 1997). Piscivorous birds display a number of symptoms similar to those observed in other avian species exposed to planar halogenated hydrocarbons in the laboratory including altered biochemical homeostasis, physical deformities, fetotoxicity, and teratogenesis. In addition to embryo mortality, PCBs cause edema and beak malformations often recognized as crossed beaks in double-crested cormorants.

Wildlife. Wildlife, especially mink, are particularly susceptible to adverse effects from exposure to specific PCB congeners, including the non-ortho and mono-ortho substituted

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PCBs, because their mechanism of action is similar to 2,3,7,8-TCDD (Leonards et al., 1994). Residues from PCBs can cause mortality or serious reproductive complications in mammals. Other clinical signs of PCB toxicity include anorexia, liver and kidney degeneration, and gastric ulcers, which have been observed in mink fed PCB-contaminated fish (Wren, 1987a,b).

2.5.3 Pesticides

Organochlorine pesticides include the chlorinated ethane derivatives, such as DDT; cyclodiene compounds, which include chlordanen aldrin, dieldrin, heptachlor, endrin, and toxaphene; and the hexachlorocyclohexanes, such as lindane. This class of compounds is persistent in the environment and possesses a strong tendency to bioaccumulate in biological tissue. Organochlorine pesticides exhibit a wide range of acute toxicities.

2.5.3.1 Mode of Action

As neurotoxins, the target locus of primary toxic action of at least some organochlorine pesticides is believed to be sensory and motor nerve fibers and the motor cortex in vertebrates (Klaassen et al., 1986). Research has implicated the role of these compounds in inhibiting neurological ion transport as an explanation for the observed toxicological effects.

2.5.3.2 Ecotoxicology of Specific Taxonomic Groups

Invertebrates. The organochlorine pesticide heptachlor is highly toxic to aquatic invertebrates with 96-hour LC_{50}s reported as less than 1 ug/L.

Fish. Fish are relatively sensitive to organochlorine pesticides with 96 hour LC_{50}s for dieldrin, and gamma BHC on the order of 1-10 ug/L.

Birds. In general, birds are less sensitive to organochlorine pesticide exposure than are aquatic organisms. However, there has much concern over chronic exposure of bird species to DDT, DDE, and gamma BHC and effects on reproduction, particularly eggshell thinning and embryotoxicity. Laboratory studies have demonstrated the potential for DDT and DDE to cause subtle changes in courtship behavior, delays in pairing and egg lying, and decreases in egg weight.

2.5.4 Polycyclic Aromatic Hydrocarbons

PAHs are a class of nonpolar organic contaminants characterized by their highly aromatic fused ring structures. They contain two to seven fused aromatic rings consisting of carbon and hydrogen (Douben, 2003). PAHs form as by-products from the incomplete combustion of organic materials. Environmental sources of PAHs include petroleum products and combustion residues. Because of their low aqueous solubilities, high

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octanol-water partition coefficients (log K_{ow} = 3.4 to 7.6), and relatively low volatilities, PAH compounds in aquatic systems tend to be associated with sediments and biota. The fate and transport of PAHs is largely controlled by sediment organic carbon content and dissolved organic carbon concentrations in the water column and sediment porewater. The extent to which an individual PAH compound will tend to be associated with either sediment or dissolved organic carbon, depends on the relative hydrophobicity of the compound. Most PAHs introduced to aquatic systems will settle to the sediments with only very low concentrations of PAHs being found dissolved in overlying waters (Neff, 1979; Maliszewska-Kordybach, 1999).

PAHs are degraded by photochemical reactions, microbial activity, and by metabolism in higher organisms (Douben, 2003). Lower molecular weight PAHs (LPAHs, comprised of three aromatic rings or less) are more water soluble and more easily degraded. Higher molecular weight compounds (HPAHs) will tend to predominate in sediments where they are subjected to burial, resuspension, and degradation reactions. The available literature suggests that higher molecular weight PAHs are degraded by microbes slower than lower molecular weight PAHs.

Uptake of PAH compounds by aquatic biota is rapid. However, PAH compounds are also quickly metabolized and eliminated by most fish species and the bioavailability of PAHs in sediments tends to decrease with time (Leppanen and Kukkonen, 2000). Invertebrates, especially mollusks, do not metabolize PAHs as efficiently and may accumulate high tissue concentrations (Eisler 1987). Bioconcentration factors for those species that do not metabolize PAHs tend to increase as the molecular weight and the octanol-water partition coefficients of the PAHs increase. Organisms with higher lipid contents bioaccumulate PAHs to a greater extent, and the rate and level of PAH uptake is greater for organisms feeding at or below sediment surfaces, including deposit feeders, browers, and meiobenthic copepods (Burgess et al., 2003). Biomagnification of PAHs does not appear to occur in aquatic and terrestrial systems due to biotransformation of PAHs by vertebrates and some invertebrates (Den Besten et al., 2003).

2.5.4.1 Mode of Action

Narcosis. PAHs are considered to be Type I narcotic compounds, with exposure resulting in a general suppression of various physiological processes in biological systems (Verhaar et al., 1992).

Phototoxicity. UV radiation can greatly enhance the toxic effects of some PAHs (Ankley, et al., 2003). It is thought that PAHs that exhibit phototoxicological properties can generate superoxide anion radicals in the presence of near ultraviolet light. In the absence of oxygen, these radicals act as photoreducing agents and can cause non-specific tissue damage (INCHEM, 1998). UV-enhanced toxicity of PAHs has been demonstrated

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for a number of organisms including mammals, fish, crustaceans, mollusks, annelids, amphibians, insects, and plants (Ankley et al., 2003).

DNA Adduct Formation. PAHs, such as benzo[a]pyrene may form adducts with DNA, potentially resulting in mutations and cancer development. Previous studies have found correlations between PAH-DNA adducts in flatfish, including European eel and English sole, and PAH concentrations in sediment (Van der Oost et al., 1994; French et al., 1996).

2.5.4.2 Ecotoxicology of Specific Taxonomic Groups

PAHs vary considerably in their toxicity to organisms. LPAHs (including naphthalene, fluorine, phenanthrene, and anthracene) are acutely toxic to aquatic organisms; this toxicity increases with increasing alkyl substitution (Van Luik, 1984). Many of the HPAHs (including chrysene and benzo(a)pyrene) are less acutely toxic but are carcinogenic, mutagenic, or teratogenic to a wide variety of organisms including fish, amphibians, birds, and mammals (Eisler, 1987). Among aquatic organisms, acute toxicity is most pronounced among crustaceans and least pronounced among teleosts (Eisler, 1987).

The bioaccumulation of several PAH compounds (including anthracene, pyrene, benzo(a)anthracene, chrysene, perylene, and benzo(a)pyrene) has also been shown to be acutely toxic to both aquatic invertebrates and fish species when the contaminated organisms are exposed to sunlight or ultraviolet radiation (Newsted and Giesy, 1987). Little or no data exist on the uptake rates of PAHs by terrestrial vertebrates (Malcolm and Shore, 2003).

Benthic Invertebrates. Effects of PAHs observed in benthic invertebrates include inhibited reproduction, delayed emergences, sediment avoidance, and mortality (Eisler, 1987; Landrum et al., 1991). Aquatic invertebrates are less able to metabolize PAHs than aquatic vertebrates, although capabilities vary widely within and among taxa (Meador et al., 1995).

Fish. PAHs are generally hydrophobic compounds and must be metabolized to more water-soluble forms before excretion can occur. In most fish species, PAHs are rapidly absorbed, metabolized, and excreted so that tissue residue concentrations are typically low. Fish exposed to PAHs may be induced to produce higher levels of mixed-function oxidase (MFO) enzymes capable of transforming PAHs to more excretable (and sometimes more toxic) metabolites (O'Connor and Huggett, 1988).

PAH contaminated sediment has been linked to adverse effects in fish, including reproductive impairment, immune dysfunction, increased incidence of liver lesions and other histopathological endpoints (Baumann, 1996; Eisler, 1987). Fin erosion and liver abnormalities have been observed in fish exposed to extracts from PAH-contaminated

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sediments (Fabacher et al., 1991). Other studies report sublethal effects on the cellular immune system in fish exposed to PAH contaminated sediment, which could result in increased susceptibility to disease (Weeks et al., 1986). PAH contaminated sediments have been associated with hepatic tumors in fish and certain PAHs can cause physiological changes and affect growth, reproduction, swimming performance, and respiration. The most common diseases generally affect the liver, although cataracts and pollution-related disorders of the skin and gills may also occur (O'Connor and Huggett, 1988).

Birds. PAHs are rapidly metabolized by birds, with metabolism of PAHs also occurring in the egg (Malcolm and Shore, 2003). Acenaphthene, fluorene, anthracene, and phenanthrene were found to be acutely toxic (LD50) to red-winged blackbirds (*Agelaius phoeniceus*) exposed via gavage at concentrations ranging from 101 to 113 mg/kg body weight (Schafer et al., 1983). Patton and Dieter (1980) fed mallards diets that contained 4,000 mg PAHs/kg for a period of 7 months. No mortality or evidence of toxicity was noted, however, the duck's liver weights increased 25 percent and blood flow to the liver increased 30 percent when compared to controls (Eisler, 1987). PAHs have been shown to cause reproductive toxicity in birds, although most studies from available literature involve exposure of PAHs to adult birds or eggs via injection or topical application (for eggs) (Malcolm and Shore, 2003). PAH mixtures have been shown to increase embryo mortality and deformation when applied to the surface of mallard eggs (Hoffman and Gay, 1981). Complete infertility occurred in female pigeons injected on a weekly basis with 10 mg/kg body weight benzo[a]pyrene over a period of up to 5 months, which was believed to be due to changes in estrogen levels (Hough et al., 1993).

Mammals. PAH exposure has been shown to result in acute toxicity and reproductive toxicity in laboratory mammals (Malcolm and Shore, 2003). Oral LD_{50}s for mice and rats exposed to naphthalene are approximately 100-500 and 2700 mg/kg body weight, respectively. Benzo(a)anthracene, benzo(a)pyrene, dibenzo(a,h)anthracene, and naphthalene have also been shown to be embryotoxic to mice and rats.

2.5.5 Volatile Organic Compounds (VOCs)

Volatilization and biodegradation are the major fate processes affecting VOC compounds in aquatic environments. In general, VOCs are not bioconcentrated by aquatic organisms and are not bioaccumulated by mammals and birds. VOCs will exist primarily in the aqueous compartment rather than sediment, due to characteristically high solubilities and low sediment adsorption coefficients. Most VOCs are considered to effect biological organisms through a narcotic mode of action, which results in a general suppression of various physiological processes (Verhaar et al., 1992).

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2.5.6 Toxic Metals

Key factors that affect the partitioning and speciation, and thus the bioavailability, of sediment metals include Eh (redox potential), pH, porewater hardness, and the organic carbon content of the sediment. The redox conditions, pH, and the porewater concentrations of dissolved organic carbon influence the oxidation state and the dissolved speciation of the metal. Metals exhibit a range of binding affinities with both organic and inorganic phases present in the sediment resulting in varying concentrations of dissolved versus particulate metals. In addition, metals exhibit a range of stability constants with dissolved ligands which determines the ratio of complexed to freely dissolved species in solution.

Total sediment metals concentrations are generally not predictive of the bioavailability of these trace metals. Concentrations of certain metals in porewaters have been correlated with biological effects (Di Toro et al., 1990). For several divalent metals, a key partitioning phase controlling cationic metal activity and toxicity in sediments appears to be acid volatile sulfide (AVS; DiToro et al., 1990, 1992; Carlson et al., 1991; Ankley et al., 1993). Simultaneously extracted metals (SEM) and acid volatile sulfide (AVS) measurements can be made to assess the potential bioavailablity of SEM metals (including cadmium, copper, lead, nickel, and zinc).

The bioavailability of metals that form stable complexes with organic compounds is particularly complex. Methylmercury compounds are extremely toxic and are efficiently bioaccumulated through aquatic food chains (Wiener and Spry, 1996). Methylmercury is formed in aquatic sediments due to microbial methylation of inorganic mercury.

2.5.6.1 Mode of Action

Most metals affect multiple organ systems and the targets for toxicity are specific biochemical processes, such as enzymes and/or membranes of cells and organelles. The toxicological effect is usually exerted as a result of an interaction between the free metal ion and the target organ (Klaassen et al., 1986). The ultimate determinant of metal toxicity is dependent upon the molar concentration of each inorganic species and where it is located within the organism.

2.5.6.2 Ecotoxicology of Specific Taxonomic Groups

The combination of trace elements in the environment may result in additive, synergistic, or antagonistic effects, with the overall effect depending on the toxicity of the metals in question, the specific physical and chemical conditions of the site, and internal synergistic or antagonistic effects within organisms. In some species, certain trace elements are physiologically regulated to levels above ambient environmental conditions (e.g., copper used in respiratory enzymes).

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Invertebrates. Toxicity of trace elements to invertebrates ranges widely, from slight reduction in growth rates to acute mortality. Mollusks are generally less sensitive than other aquatic phyla (Leland and Kuwabara, 1985). The most sensitive life stages of benthic organisms are generally the embryonic and larval stages. Arsenic, cadmium, chromium, lead, and zinc have all been shown to reduce survival and cause reproductive impairment in benthic invertebrates. Reduced growth is often observed in invertebrates at sublethal exposures to cadmium, copper, lead, mercury, nickel, silver, and zinc. In addition, aquatic exposure to nickel, silver, and zinc can result in abnormal development and cadmium exposure can result in molt inhibition at low ambient concentrations.

Fish. Freshwater fish are generally more sensitive to the effects of inorganics than are marine species, and the larval stages are generally more sensitive. Commonly observed effects include retarded growth, mortality, and reduced fecundity.

Birds. Avian dietary toxicity studies have been conducted with a wide range of inorganics. The observed acute toxicity often depends on the organism's level of metallothioneins. Ducks contained the highest levels of metallothioneins of a range of wildlife species surveyed (Eisler, 1985). Sublethal effects can include reproductive and behavioral modifications. Methylmercury is one of the most toxic trace elements; immersion of mallard eggs in solutions of this contaminant resulted in a significant incidence of skeletal embryonic malformations (Eisler, 1985).

2.6 ASSESSMENT AND MEASUREMENT ENDPOINTS

As the culmination of the Problem Formulation phase of the BERA, endpoints were identified in order to assess the risks posed by COPC exposure to ecological receptors at the Site. Assessment endpoints are explicit expressions of the environmental values (e.g., ecological resources) that are to be protected. Four principal criteria are used to select ecological values that may be appropriate for assessment endpoints: 1) ecological relevance, 2) susceptibility to known or potential stressors, 3) commercial or social value, and 4) relevance to management goals (USEPA, 1998). Adverse risks to assessment endpoints drive any potentially necessary risk management decisions. Assessment endpoints generally are populations, communities, or trophic guilds (e.g., insectivorous birds). Populations or trophic guilds may be deemed at risk if reproduction or survival of individuals is determined to be significantly impacted. The general types of effects of concern include:

- mortality, growth, or reproductive effects resulting from direct exposure to contaminants that affect a significant proportion of a receptor population;
- mortality, growth, or reproductive effects resulting from exposure to contaminants that have bioaccumulated in the ecological food chain that affect a significant proportion of a (higher trophic level) receptor population; and,

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- indirect effects associated with a substantial reduction in abundance of prey populations.

The assessment endpoints selected for the BERA consist of the following:

1. Protection and maintenance (i.e., survival, growth, reproduction) of aquatic and floodplain invertebrate communities which are a forage base for fish and wildlife.
2. Protection and maintenance of demersal, omnivorous fish populations as a forage base or sport fishery.
3. Protection and maintenance of pelagic, piscivorous or semi-piscivorous fish populations as a forage base or sport fishery.
4. Protection and maintenance of piscivorous mammal and bird populations.
5. Protection and maintenance of insectivorous mammal and bird populations.
6. Protection and maintenance of omnivorous mammal and bird populations.

Assessment Endpoint 1 was segregated into two separate components (i.e., 1a and 1b; focused on the aquatic and floodplain invertebrates communities, respectively) to facilitate the risk analysis (Table 15). For Assessment Endpoint 5, the bird and mammal species that forage on flying insects (true insectivores) was distinguished from those that feed primarily on earthworms (i.e., vermivores) in Section 8.

Measurement endpoints are quantifiable ecological characteristics, through laboratory or field experimentation, that are related to the valued characteristic chosen as the assessment endpoint (USEPA, 1992, 1998). The measurement endpoint should be sensitive, and represent the same exposure pathway and mechanisms of toxicity as the assessment endpoint it represents. Types of measurement endpoints used in the risk assessment process fall generally into three categories: 1) comparison of estimated or measured exposure levels of COPCs to levels known to cause adverse effects, 2) bioassay testing of site and background media, and 3) comparison of observed population- and community-level effects in areas downgradient of the source area with those observed at background or reference areas. Measurement endpoints selected for assessment endpoint evaluation in this risk assessment included data obtained from all three categories. The selected assessment and measurement endpoints and the *a priori* scoring for the weight of evidence evaluation are summarized in Table 15. Each measurement endpoint was assigned a relative ranking ("overall score") based on assignments for 10 separate attributes that are related to the utility of that endpoint to answer the risk questions and resolve the assessment endpoint (MWOEW, 1995). Overall scores range from "low" to "high" and endpoints with higher *a priori* determined rankings are given greater inference weight in the risk characterization process.

Tables 16 and 17 summarize the endpoint receptors and evaluated wildlife exposure pathways, respectively. Tables 18 through 20 present the food web modeling inputs for piscivorous, insectivorous, and omnivorous wildlife receptors, respectively, and

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equations used to model wildlife exposures are provided in Tables 21 and 22 for aquatic and floodplain exposures, respectively.

The specific BERA assessment and measurement endpoint are described in the following section.

2.6.1.1 Aquatic Invertebrate Community

Assessment Endpoint. Protection and maintenance (i.e., survival, growth, reproduction) of aquatic invertebrate communities, which are a forage base for fish and wildlife.

The following risk questions were established for this assessment endpoint:

A. Do measured concentrations of COPCs in surface water exceed appropriate criteria and/or guidelines for the protection of aquatic life, with special consideration of reproduction and early lifestage survival?
B. Do measured concentrations of COPCs in whole sediment exceed appropriate guidelines for the protection of benthic macroinvertebrate populations?
C. Are sediments toxic to aquatic macroinvertebrates under chronic laboratory exposure conditions?
D. Do measured and modeled concentrations of COPCs in the tissues of aquatic macroinvertebrates (such as aquatic insects, crayfish, and molluscs) exceed benchmarks for residue effects on survival, growth or reproduction?
E. Do the available aquatic macroinvertebrate data indicate presence/absence of ecological integrity due to VOC discharge in riffle/run habitat adjacent to the Site?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of surface water COPC concentrations to criteria/guidelines.
B. Comparison of sediment COPC concentrations to benchmarks/guidelines.
C. Site-specific whole sediment laboratory bioassays.
D. Comparison of measured COPC concentrations in aquatic macroinvertebrates to CBRs.
E_1. Site-specific study of aquatic macroinvertebrate community structure/function.
E_2. Site-specific study of emerging aquatic macroinvertebrate productivity.

A weight of evidence approach was used to assess the health of the aquatic benthic macroinvertebrate community. Sediment laboratory bioassays, a benthic community structure assessment, an assessment of the productivity of the emerging insect fauna, and comparison of surface water, sediment, and tissue residue chemistry to Toxicity

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Reference Values (TRVs) provided measures of exposure and effect that were integrated in the analysis.

Selected TRVs were compared to surface water and sediment exposure point concentrations (EPCs) for each exposure area, and sediment benchmarks were used to analyze the pattern of laboratory bioassay response in two species of aquatic invertebrates. In addition, crayfish, emerging insect, and tree swallow stomach content (i.e., captured insects comprising the diet of tree swallow nestlings) tissue samples were collected and the analytical tissue chemistry results were used as an indicator of exposure to aquatic invertebrates (and indirectly to wildlife consumer species). Critical body residue (CBR) threshold concentrations were identified and compared to the tissue chemistry data to determine whether contaminant uptake poses a significant bioaccumulation hazard to aquatic organisms. Two community-level measurement endpoints were evaluated in the BERA; a macroinvertebrate community assessment conducted in riverine sections of the study area and an assessment of the productivity of the emerging insect community associated with depositional lacustrine habitat. The sediment bioassays were also conducted using sediments from depositional areas.

As discussed in Section 2.1, the Woonasquatucket River provides both riverine (lotic) and lacustrine (lentic) habitat, and the specific aquatic receptors and COPC exposures likely differ between these two areas. The benthic community study using the USEPA Rapid Bioassessment Protocol was conducted to assess potential community level effects in the lotic reach of the Woonasquatucket River adjacent to the Site. The laboratory sediment bioassays were conducted using sediment collected from depositional areas throughout the study area.

Sampling locations for the benthic macroinvertebrate community endpoint (i.e., E_1) were chosen to represent a range of concentrations for the COPCs that are most likely to adversely affect aquatic invertebrates (including PAHs, pesticides/PCBs, and inorganics). Although, exposure to VOCs is not generally a substantial concern for aquatic invertebrates, several sampling locations for the macroinvertebrate community assessment were located in the reach of the river adjacent to the Centredale Manor source area, where VOC concentrations in discharging overburden groundwater are elevated. Based on the results of the vapor diffusion sampling, MACTEC selected four RBP locations along the Woonasquatucket River. WRC-SW-4001 is located approximately 100 feet south of the U.S. Route 44 bridge, and represents upstream background conditions. WRC-SW-4002 is located approximately 400 feet downstream of the bridge and is located in the vicinity of the northern (most upstream) bounds of the VOC-impacted area (i.e., where PCE was first detected). WRC-SW-4003 is located west of Cap #1, and downstream of the area where the highest concentrations of TCE and PCE were detected in the water-to-vapor diffusion samples. WRC-SW-4004 is located approximately 1800 feet downstream of the U.S. Route 44 bridge, where secondarily elevated levels of PCE were detected.

In general, site-specific measures of effect (i.e., endpoints C and E_1) were assigned a higher ("medium-high") *a priori* scoring than those based on comparison of literature-based TRVs to abiotic media (Table 15). This reflects the greater value of site-specific studies, which are conducted at higher level of ecological organization, than represented by standard benchmarks. The aquatic invertebrate risk evaluation is presented in Section 3.

2.6.1.2 Floodplain Invertebrate Community

Assessment Endpoint. Protection and maintenance (i.e., survival, growth, reproduction) of floodplain invertebrate communities, which are a forage base for wildlife.

The following risk questions were established for this assessment endpoint:

A. Do measured concentrations of COPCs in floodplain soil exceed appropriate guidelines for the protection of floodplain soil invertebrate populations?
B. Do measured and modeled concentrations of COPCs in the tissues of floodplain soil invertebrates (such as earthworms) exceed benchmarks for residue effects on survival, growth or reproduction?
C. Do the available floodplain soil invertebrate data indicate presence/absence of ecological integrity?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of floodplain soil COPC concentrations to benchmarks/guidelines.
B. Comparison of measured site-specific COPC concentrations in floodplain soil invertebrates to CBRs.
C. Site-specific study of floodplain soil invertebrate community structure/function.

Measurement endpoints used to assess the health of the floodplain invertebrate community included a community level field study and comparisons of surface soil and tissue residue chemistry to Toxicity Reference Values (TRVs).

Selected literature-derived TRVs for soil earthworms were compared to surface soil EPCs for Allendale and Lyman Mill. In addition, earthworm tissue samples were collected and the analytical tissue chemistry results were used as to medel soil contaminant exposure to floodplain invertebrates (and indirectly to wildlife consumer species). CBR threshold concentrations were identified and compared to the tissue chemistry data to determine whether contaminant uptake poses a significant bioaccumulation hazard to soil invertebrates. Measures of community integrity determined at the Allendale and Lyman Mill EAs were compared to results obtained from

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the upstream background area to draw conclusions about the health of this endpoint receptor within the study area.

Measurement endpoints A and B were assigned a "low-medium" *a priori* score due to the uncertainties in extrapolating effects from literature studies (for which contaminant availability and species sensitivity may differ from site conditions). The community level field study (i.e., endpoint C) was assigned a "medium" *a priori* score, a consequence of the fact that this study and the assessment point focus on the same high level of ecological organization (i.e., community level). The floodplain invertebrate risk evaluation is presented in Section 4.

2.6.1.3 Demersal Fish Populations

Assessment Endpoint. Protection and maintenance of demersal, omnivorous fish populations as a forage base or sport fishery.

The following risk questions were established for this assessment endpoint:

A. Do field surveys of demersal fish populations indicate presence/absence of ecological integrity?
B. Do measured and modeled concentrations of COPCs or TEQs in eggs and/or tissues of demersal, omnivorous fish (such as carps, suckers, other cyprinids, bullheads, turtles or American eel) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A_1. Fish length-weight relationships and condition indices relative to habitat characterization.
A_2. Percent gross lesions in individual fish.
A_3. Demographic structure analysis of dominant species.
A_4. Species richness and relative abundance of ichthyoplankton.
B_1. Comparison of measured concentrations or toxic equivalencies in fish tissue to literature derived CBRs.
B_2. Comparison of modeled concentrations or toxic equivalencies in eggs and fish tissue to site-specific CBRs.
B_3. Partial life cycle laboratory bioassay using channel catfish eggs, embryos, and fry.

Potential effects of COPCs on demersal fish populations were evaluated by comparing COPC concentrations measured in analyzed whole body fish tissue to CBRs. Measures of effect derived from a fish community assessment field study included length-weight relationships for individual fish, a population demographic analysis, the calculation of

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condition indices based on mature fish for separate EAs, an evaluation of the incidence of gross lesions in individual fish, and an evaluation of the species richness and relative abundance of immature fish and egg (i.e., ichthyoplankton) in each EA.

During the study design process, it was determined that the existing fish tissue chemistry data were not adequate to develop EPCs for the BERA due to the chemical parameters analyzed, the analytical method detection limits, the spatial coverage represented by the samples and the historical nature of these data. Consequently, analytical tissue chemistry for three fish species (including two demersal species) was collected during the 2001 field sampling event. A quantitative fish population bioassessment (including ichthyoplankton) study was also conducted because the existing information was not of sufficient scope or coverage within the EAs. Finally, the presence of HCX at elevated concentrations in study area sediments and the inadequate scientific understanding of the toxicity of this compound (and possible interactions with other COPCs) led to the development of an early-life stage fish egg bioassay to address this uncertainty. The bioassay and the comparison of effect thresholds derived from the study to measured fish tissue concentrations were included as two additional measurement endpoints for this evaluation.

In general, the measurement endpoints associated with the fish community field study were assigned "low to medium" *a priori* scores (Table 15); although linked to the assessment endpoint, these endpoints are of poor sensitivity and lack a clear stressor/response relationship. The comparison of measured fish tissue concentrations to literature-derived or bioassay-derived CBRs were considered to provide a strong indication of exposure and this endpoint and ELS study (i.e., endpoints B_1, B_2, and C) were assigned "medium" or "medium-high" *a priori* scores. The risk evaluation for demersal fish populations is presented in Section 5.

2.6.1.4 Pelagic Fish Populations

Assessment Endpoint. Protection and maintenance of pelagic, piscivorous or semi-piscivorous fish populations as a forage base or sport fishery.

The following risk questions were established for this assessment endpoint:

A. Do field surveys of piscivorous fish populations indicate presence/absence of ecological integrity?
B. Do measured and modeled concentrations of COPCs or TEQs in eggs and/or tissues of pelagic, piscivorous or semi-piscivorous fish (such as sunfishes, other centrarchids, or yellow perch) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?

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The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A_1. Fish length-weight relationships and condition indices relative to habitat characterization.
A_2. Percent gross lesions in individual fish.
A_3. Demographic structure analysis of dominant species.
A_4. Species richness and relative abundance of ichthyoplankton.
B_1. Comparison of measured concentrations or toxic equivalencies in fish tissue to literature derived CBRs.
B_2. Comparison of modeled concentrations or toxic equivalencies in eggs and fish tissue to site-specific CBRs.

The study design for the pelagic fish population endpoint parallels that discussed for the demersal fish evaluation. However, the ELS bioassay was not designed specifically to assess this endpoint.

2.6.1.5 Piscivorous Wildlife Populations

Assessment Endpoint. Protection and maintenance of piscivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

A. Do ingestion doses of COPCs in piscivorous wildlife (such as mink, river otter, kingfisher, great blue heron, or American bald eagle) exceed Toxic Reference Values (TRVs) or Toxic Equivalencies (TEQs) for adverse effects on survival, growth or reproduction?
B. Do residues of COPCs in tissues of piscivorous wildlife (such as mink, river otter, kingfisher, great blue heron, or American bald eagle) exceed benchmarks for adverse effects on survival, growth, or reproduction?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of estimated ingestion doses in piscivorous wildlife with TRVs and toxic equivalencies.
B. Comparison of estimated piscivorous wildlife residues with CBRs.

Risks to piscivorous wildlife populations were evaluated by comparing estimated dietary doses (associated with consumption of contaminated drinking water, sediment and fish) for kingfisher, great blue heron and river otter to TRVs and TCDD TEQs, and comparing

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estimated wildlife tissue (including avian egg) data to CBRs (Table 15). Exposure modeling was conducted using the modeling inputs presented in Table 18.

During the study design process, it was determined that the available biological tissue data were inadequate to reliably estimate exposure doses to piscivorous wildlife receptors and analytical tissue chemistry for three fish species was collected during the 2001 field sampling event. The white sucker was preferentially used to estimate dietary exposures to piscivores associated with consuming contaminated fish because of its intimate association with bed sediment. However, this species was not collected in Assapumpset, Manton, or Dyerville during the 2001 field sampling program. Brown bullhead and largemouth bass whole body tissue data were used to derive EPCs for fish tissue in Assapumpset and Manton, respectively. Only American eel was collected in Dyerville and for consistency in exposure modeling, biota-sediment accumulation factors (BSAFs) were developed using the available sediment and white sucker chemistry data and were used to estimate dietary exposures to piscivorous wildlife receptors in this EA.

Of the two endpoints, endpoint A was assigned a slightly higher ranking ("medium" vs "low-medium") due primarily to the use of measured, rather than estimated exposure concentrations. The piscivorous risk evaluation is presented in Section 7.

2.6.1.6 Insectivorous Wildlife Populations

Assessment Endpoint. Protection and maintenance of insectivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

E. Do ingestion doses of COPCs in insectivorous wildlife (such as shrew, bat, American woodcock or tree swallow) exceed Toxic Reference Values (TRVs) or Toxic Equivalencies (TEQs) for adverse effects on survival, growth or reproduction?
F. Do residues of COPCs in eggs and/or tissues of insectivorous wildlife (such as shrew, Indiana bat, American woodcock or tree swallow) exceed benchmarks for adverse effects on survival, growth, reproduction or embryo development?
G. Do field survey data on insectivorous wildlife population indicate presence/absence of ecological integrity?
H. Do mix-function oxidase (MFO) liver enzyme levels in swallow nestlings indicate that they have been exposed to compounds with "dioxin"-like metabolic activity?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

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A. Comparison of estimated ingestion doses in insectivorous wildlife with TRVs and toxic equivalencies.
B_1. Comparison of measured insectivorous wildlife tissue and/or egg residues with CBR data.
B_2 Comparison of estimated insectivorous wildlife tissue and/or egg residues with site-specific CBR data.
C_1 Site-specific measurement of reproductive effects in local tree swallow populations.
C_2 Site-specific survey of calling amphibians.
D. Elevated MFO activity in tree swallow nestling liver tissue.

Risks to insectivorous species were evaluated by comparing estimated dietary doses (associated with consumption of contaminated drinking water and emerging insect prey) for adult and nestling tree swallows and little brown bat to TRVs and TCDD TEQs; comparing measured and estimated wildlife tissue (including avian egg) data to CBRs, measuring reproductive effects in local tree swallow populations during a four year field study, and by conducting qualitative surveys of indigenous wildlife populations. In addition to the field study, dietary exposures to adult (and nestling) swallows were modeled because the focus of the study was on reproductive effects on the developing embryos. The dietary exposure modeling for the tree swallow nestlings was done to supplement the limited field effects data for this life stage and because the USGS study did not study fledgling survival.

The same measurement endpoints were used to estimate potential risk to vermivorous species, with the exception that no reproductive study was conducted for vermivorous species. Dietary exposures to the woodcock and short-tailed shrew were estimated using contaminant concentrations in measured in floodplain soil and earthworms. Contaminant exposures in plant diet of the shrew were estimated using literature-derived uptake factors. Exposure modeling was conducted using the modeling inputs presented in Table 19.

During the study design process, it was determined that the available biological tissue data were inadequate to reliably estimate exposure doses to vermivorous wildlife receptors and chemical analysis of paired earthworm tissue and floodplain soil samples were collected during the 2001 field sampling event. In addition, chemical analysis of emerging insect tissue was conducted during 2001 and of stomach contents of tree swallow nestlings was conducted in 2000, 2001, and 2003 to support exposure modeling to insectivorous wildlife.

Floodplain soil chemistry data are not available for either Manton or Dyerville. During the study design process, it was concluded that there was a low likelihood of significant contaminant transport to floodplain habitats downstream of Lyman Mill; this is supported by the distributional analysis discussed in Section 1.7. In addition, there is relatively

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little floodplain habitat in this lower reach of the study area (Figure 5). As a result, potential risks to vermivorous wildlife (i.e., the woodcock and short-tailed shrew endpoint receptors) were only evaluated in Allendale, Lyman Mill, and Greystone, the upriver background area. The collection of emerging insect tissue for chemical analysis was limited to the areas included in the multi-year tree swallow study and data were not collected in the Dyerville EA. However, the presence of elevated COPC concentrations in sediment samples from this EAs suggests that a potential bioaccumulation hazard exists for insectivorous birds and mammals in the lower portion of the study area. Consequently, BSAFs were developed using the available sediment, emerging insect and swallow stomach content chemistry data and used to estimate dietary exposures to aerial foraging insectivores in both Dyerville and Manton.

During the study design process, it was determined that the available biological tissue data were inadequate to reliably estimate exposure doses to vermivorous wildlife receptors and chemical analysis of paired earthworm tissue and floodplain soil samples and paired crayfish and composite sediment samples were collected during the 2001 field sampling event.

The three assessment endpoints that involved estimating ingestion doses, tissue residue concentrations, and based on measured EROD activity in tree swallow nestling liver tissue were assigned a "medium" *a priori* score. The tree swallow reproductive study and the comparison of measured swallow egg TCDD TEQs to CBRs were assigned a "high" and "medium-high" *a priori* rankings, respectively. The anuran survey (i.e., endpoint C_2) was assigned a "low-medium" *a priori* score due to the relatively poor sensitivity and qualitative nature of this study. The insectivorous risk evaluation is presented in Section 8.

2.6.1.7 Omnivorous Wildlife Populations

Assessment Endpoint. Protection and maintenance of omnivorous mammal and bird populations.

The following risk questions were established for this assessment endpoint:

A. Do ingestion doses of COPCs in omnivorous wildlife (such as raccoon or mallard) exceed Toxic Reference Values (TRVs) or Toxic Equivalencies (TEQs) for adverse effects on survival, growth or reproduction?

The following measurement endpoints were selected to evaluate risk to this endpoint receptor group (letters correspond to the risk questions above):

A. Comparison of estimated ingestion doses in omnivorous wildlife with TRVs and toxic equivalencies.

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Risks to omnivorous receptor populations were evaluated by comparing estimated dietary doses (associated with consumption of contaminated drinking water and prey and the incidental ingestion of sediment/soil) for mallards and raccoons to TRVs and TCDD TEQs. It was assumed that the raccoon could be exposed to contaminants associated with both aquatic and floodplain habitat. Exposure modeling was conducted using the modeling inputs presented in Table 20.

During the study design process, it was determined that the available biological tissue data were inadequate to reliably estimate exposure doses to omnivorous wildlife receptors and chemical analysis of paired earthworm tissue and floodplain soil samples and paired crayfish and composite sediment samples were collected during the 2001 field sampling event.

The sole measurement endpoint used to evaluate risks to omnivorous wildlife was assigned a "medium" *a priori* score. The omnivorous risk evaluation is presented in Section 9.

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SECTION 3.0

3.0 AQUATIC INVERTEBRATE COMMUNITY RISK EVALUATION

This section evaluates risks to aquatic invertebrates using the measurement endpoints discussed in Section 2. The evaluation of the floodplain soil invertebrate community is presented in Section 4. The measurement endpoints, sampling strategy, and methods are described, followed by an evaluation of surface water, sediment, and biological tissue. Measured effects on benthic invertebrates are evaluated based on the results of toxicity tests and the macroinvertebrate community study. The results from the analytical chemistry, toxicity bioassays, and community analyses are then integrated to characterize risks to this community. As discussed in Section 2.6.1.1, the macroinvertebrate community study and sediment bioassays were specifically selected to evaluate exposures in separate aquatic habitats within the study area (i.e., lotic and lentic, respectively).

Further details of study design rationale and methods are presented in the Final Work Plan and the Final FSP (FSP, Harding ESE, 2001a; 2001b).

3.1 EXPOSURE ASSESSMENT - AQUATIC INVERTEBRATES

In this section, COPC exposure by aquatic invertebrates is assessed using both historical and additional analytical data collected in 2001. Table A-1 summarizes the information on the analytical samples for surface water, groundwater, sediment, crayfish, and emerging insects that were used in this evaluation. These data were collected throughout the study area to represent a range of exposure to different contaminants and to integrate abiotic media concentrations with toxicity and macroinvertebrate community structure measured in co-located samples. In addition, analytical chemistry data for biota tissue and associated sediment samples were used to develop BSAFs for predictive modeling and deriving PRGs.

3.1.1 Exposure Concentration Estimation

This section describes the levels of COPCs found in the various environmental media at the Site. EPCs were developed from the Site database, which was queried to provide the following for each COPC, medium, and exposure area:

- Total number of samples
- Frequency of detection
- Maximum detected concentration
- Minimum detected concentration
- Arithmetic mean concentration
- Data distribution (normal, log-normal, non-parametric)
- 95% upper confidence limit (UCL) on the mean concentration, and

- RME concentration

The central tendency (CT) concentration is the average concentration of a COPC in a given exposure area. The reasonable maximum exposure (RME) concentration represents the highest concentration of a COPC the would reasonably be expected to occur in a given exposure area; the RME concentration is calculated as either the 95th Upper Confidence Level (UCL) on the arithmetic mean or the maximum detected concentration (whichever is lower).

3.1.1.1 Quantifying Toxic Equivalencies (TEQ)

Dioxin and PCB congeners detected in Site media were expressed in terms of toxic equivalencies (TEQs) as discussed in Section 2. Congener-specific concentrations detected in all media were multiplied by Toxic Equivalency Factors (TEFs) for birds and wildlife and the TEQs were summarized according the general protocols for data reduction discussed above. Congener-specific concentrations detected in fish tissue were also multiplied by TEFs for fish (Van den Berg, et al., 1998) to compare with CBRs (Appendix G). TEQs were expressed in terms of a total 2,3,7,8-TCDD TEQ (i.e., "2,3,7,8-TCDD Toxic Equivalency") as well as the TEQs for measured dioxin/furan and PCB congeners, separately in order to identify the relative contribution of dioxin/furans and PCB congeners on the overall TEQ. The TEQ concentrations for dioxin/furans and PCB congeners are specified as "TEQ dioxin/furans" and "TEQ PCBs", respectively, with the receptor type specified (i.e., bird, mammal, fish).

The following sections describe the development of EPCs for each medium evaluated in the BERA. Each EPC table presents the arithmetic mean, the 95th percentile on the mean, the value selected to represent the RME and CT EPCs, and the dataset distribution assumptions for each COPC.

3.1.1.2 Biota-Sediment Accumulation Factors

Site-specific BSAFs (kg $_{organic\ carbon}$/kg $_{lipid}$) were calculated for organic COPCs as the ratio of the lipid-normalized COPC concentration in biota divided by the organic carbon-normalized COPC concentration in sediment. BSAFs were derived on an exposure area - and species-specific basis and incorporate the effects of metabolism, biomagnification, growth and bioavailability (USEPA, 1995). Crayfish BSAFs are presented in Tables J-2 and J-3 (Appendix J). BSAFs were calculated using the analytical chemistry for individual co-located composite crayfish tissue and sediment samples collected during the 2001 field sampling program. Analytical data for each composite crayfish and sediment sample were respectively lipid- and organic carbon-normalized and the ratio of the normalized concentrations calculated (Table J-3). BSAFs for inorganic COPCs were also calculated although an organism's lipid fraction and sediment organic carbon fraction are unlikely to be significant factors in understanding the relationship between

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tissue and sediment chemistry. Table J-2 presents the overall BSAFs, which were calculated by pooling all lipid-normalized and organic carbon-normalized data across the Site. The average, maximum, and variance of the individual BSAFs are presented along with the coefficient of variance (i.e., arithmetic mean divided by the square root of the variance estimate) for each COPC detected in crayfish tissue.

Mean BSAFs for dioxin, furan, and PCB congeners typically range from 0.2 to 0.7, although there are some exceptions (e.g., BSAFs for 1,2,3,7,8-PeCDD and 1,2,3,7,8-PeCDF are both 3.5). The TEQ BSAFs for dioxin and furan congeners are 0.65 and 0.70 based on avian and mammalian TEFs, respectively; corresponding values for the TEQ BSAFs for PCB congeners are 0.2 and 0.41 (Table J-2).

3.1.2 Surface Water/Sediment Quality Evaluation

All chemistry results as reported by the laboratory, including data qualifiers, are presented in the Final DER (Harding ESE, 2003). The laboratory data were evaluated in terms of completeness, holding times, instrument performance, accuracy, precision, method reporting limits, and field quality control samples. During the quality assurance review, no data were rejected and all data were considered useable as reported.

Analytical chemistry data for surface water and sediment were statistically summarized and compared to benchmark concentrations protective of aquatic organisms. EPCs for these media are discussed in Section 3.3. EPCs were also developed for VOCs detected in overburden groundwater that discharges to the reach of the Woonasquatucket River adjacent to Cap 1 and Cap 2.

Surface Water. Tables 23 and 24 present the surface water EPC concentrations for the CT and RME, respectively. EPCs were developed for Assapumpset, Greystone, Allendale, and Lyman Mill.

Groundwater. Tables 25 and 26 present the overburden groundwater EPC concentrations for the CT and RME, respectively. Section 2.3.6 provides a discussion of the magnitude and spatial distribution of VOCs detected using vapor diffusion samplers that were installed in the Woonasquatucket River adjacent to the Site.

Sediment. Tables 27 and 28 present the sediment EPC concentrations for the CT and RME, respectively. EPCs were developed for Assapumpset, Greystone, Allendale, Lyman Mill, Manton, and Dyerville.

3.1.3 Macroinvertebrate Tissue Residues

Macroinvertebrate tissues collected to support the BERA include crayfish, emerging insect, and tree swallow stomach content tissue. Analytical chemistry for these data were

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used to evaluate potential for direct effects to aquatic invertebrates as well as to estimate dietary exposures to wildlife receptors (Sections 7 through 9).

3.1.3.1 Crayfish Tissue

Eleven composite crayfish samples were collected at sampling stations in Assapumpset, Greystone, Allendale, and Lyman Mill and submitted for chemical analysis to determine whether tissue concentrations pose a risk of direct harm to invertebrates or to wildlife that forage on these organisms. Composite sediment samples were also collected at each of the crayfish sampling stations in order to derive BSAFs for predictive modeling of contaminant biouptake.

Three crayfish samples were collected in Allendale including CMS-CF-4003, CMS-CF-4006, and CMS-CF-4007. Sample WRL-CF-4005 was collected approximately 100 feet below the Allendale Pond Dam, where the Woonasquatucket River is free running. Sampling stations for crayfish tissue collection in Lyman Mill included LPX-CF-4004, LPX-CF-4002, and LPX-CF-4001. Sampling stations were selected based on a review of the historical sediment data in order to provide a range of potential exposures to dioxins and PCBs (Harding ESE, 2001b).

Three upstream background crayfish samples (GMP-CF-5002 [in Greystone Mill Pond], and RWR-CF-5003 and RWR-CF-5004 [in the reach of the Woonasquatucket River between Greystone and Route 44]), and one reference crayfish sample (RAB-CF-5004 in Assapumpset) were collected (Figure 58). These sample stations were located in comparable crayfish habitat and tissue levels in the samples collected upriver of the site were used to establish background levels of the analyzed compounds.

Eleven co-located crayfish tissue and sediment samples were collected between 26 July and 18 July 2001. No deviation from the SOP provided in the FSP (Harding ESE, 2001b) was encountered during the execution of this sampling program. Five or six minnow traps, with modified openings to allow passage of larger crayfish, were baited with canned cat food and deployed in the vicinity of each sampling area. Individual traps were checked routinely throughout the sampling period and crayfish were collected, placed in labeled Ziploc bags, and maintained on dry ice under chain-of-custody until the required sample mass had been obtained.

A composite sediment sample was collected at each crayfish sample station following collection of adequate biota tissue. Five separate grab samples were collected using either an Ekman or petite ponar dredge, homogenized and shipped off-site for chemical analysis. Chemical analyses included: PCB Aroclors, pesticides, metals (including methylmercury), dioxin/furan congeners, and PCB congeners. As discussed in Section 4.1.3, these analytical parameters were selected to determine whether these bioaccumulating COPCs are bioavailable (measure of exposure) and to determine

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whether concentrations exceed critical body threshold levels (measure of effect). The lipid content of the crayfish samples and the total organic carbon content of the sediment samples were also analyzed in order to derive BSAFs (Cook and Burkhard, 1999).

Crayfish Tissue. Tables 29 and 30 present the crayfish tissue EPC concentrations for the CT and RME, respectively. EPCs were developed for the Assapumpset, Greystone, Allendale, and Lyman Mill EAs.

3.1.3.2 Emerging Aquatic Macroinvertebrates Tissue

Emerging insects were sampled in Greystone, Allendale, and Lyman Mill to assess potential impacts of sediment contamination on the overall macroinvertebrate community. In addition, chemical analysis of insect tissue was conducted to determine whether tissue concentrations pose a risk of direct harm to invertebrates or to insectivorous wildlife that forage on these organisms.

The FSP indicated that five replicate emergence traps would be located in each exposure area (Figure 58). Study details are described in further detail in the work plan (Harding ESE, 2001a) and in the study report included in Appendix E.

Five aquatic insect emergence traps were deployed in the Greystone, Allendale, and Lyman Mill ponds in the general vicinity of the tree swallow population study nest boxes between 19 and 21 June 2001 (Figure 58). Traps consisted of a commercially available pyramid design structure constructed of Lumalite screening measuring 2.4 meters high and 4 square meters at the base. Traps were supported by three-fourths inch electrical conduit that was inserted onto a wooden frame which allowed the trap to float on the water surface. The partial breaching of the Allendale Pond Dam in March 2001 dramatically decreased the amount of available aquatic habitat and the emergence traps were positioned in shallow water without flotation. An additional breaching occurred during the time that the traps were deployed and the lack of precipitation resulted in water levels in Allendale Pond decreasing throughout the sample period. These five traps were repositioned twice during the sampling event; however, trap catch from this pond was minimal.

Traps were sampled every 3 to 4 days during the field program for a total of seven sampling events for each trap. Traps were sampled by opening the side of the trap and extracting emerged insects using a battery powered suction device. Insects were removed from the traps and frozen on dry ice until laboratory processing and taxonomic identification. The traps were retrieved on 24 July 2001.

At the conclusion of the sampling event, the entire sample from each replicate trap was weighed to the nearest 0.01 grams, numerical abundance of insect orders was visually determined and representative specimens of each taxon were preserved in alcohol for

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taxonomic identification. The replicate samples from Greystone and Lyman Mill were then pooled (due to the small mass of collected) and submitted for analysis of dioxin/furan and PCB congeners. The small sample volume of emerging insects that were obtained resulted in the elimination of the analysis for PCB/pesticides, metals (including methyl mercury) and lipid content as proposed in the work plan (Harding ESE, 2001a).

Tables 31 and 32 present the emerging insect tissue EPC concentrations for the CT and RME, respectively. EPCs were developed for Greystone and Lyman Mill.

As described in the tree swallow reproductive study report (included in Appendix M), the stomach contents of selected tree swallow nestlings were sampled in 2000 and 2001. Tables 33 and 34 present the tree swallow stomach contents EPC concentrations for the CT and RME, respectively. EPCs were developed for the Greystone, Allendale, Lyman Mill, and Manton.

3.2 EFFECTS ASSESSMENT - AQUATIC INVERTEBRATES

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

3.2.1 Surface Water/Sediment TRVs

TRVs used to evaluate the potential hazard associated with exposure to the surface water and sediment EPCs are documented in Tables D-1 and D-2, respectively. The basis of these TRVs is discussed in further detail in Section 2.3.1. Equilibrium partitioning sediment guidelines for metal and PAH mixtures were also derived (USEPA, 2000a; 2000b). AVS concentrations (on a molar basis) were subtracted from the sum of the molar concentrations of SEM for cadmium, copper, lead, nickel, and zinc to derive a SEM-AVS benchmark for each exposure area. It is believed that equilibrium partitioning theory applies to sediment associated silver; however, this analyte was not detected in the SEM analysis of Site sediment. USEPA (2000a) also suggests that organic carbon can influence the toxicity of these divalent cations in sediment porewater and a carbon normalized benchmark (i.e., $\Sigma SEM - AVS/f_{oc}$) value was calculated for each sample and statistically summarized for each exposure area.

PAH compounds represent the majority of the SVOC compounds detected in Woonasquatucket River sediment and Equilibrium Partitioning-based Sediment Guidelines (ESGs) were calculated for each sediment sample in which PAHs were detected and similarly summarized for each exposure area. USEPA (2000b) describes the theoretical aspects and application of this procedure to evaluating potential effects to benthic organisms associated with exposure to the overall PAH mixture in a given sediment sample. An 11.5 adjustment factor was also applied to the ESGs to estimate the

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total toxicological contribution of PAHs in Site sediment because only 17 of the 34 PAH compounds included in the ESG guidance were analyzed in sediment samples (USEPA, 2000b).

3.2.2 Critical Body Residues

Available data relating tissue residue levels in aquatic organisms to adverse ecological effects (e.g., survival, growth reproduction) were compiled from two main sources (Jarvinen and Ankley, 1999 and USACE, 2003). Jarvinen and Ankley (1999) presents tissue residue data compiled from approximately 500 separate publications and includes over 3300 records. The USACE electronic database currently contains over 3400 records compiled from 736 studies published between 1964 and 2001. The ERED database was downloaded and combined with an electronic version of the Jarvinen and Ankley dataset in a Microsoft Access database containing over 6700 records. The database was supplemented with additional information (primarily avian and mammalian) obtained from other sources including primary literature and USEPA, 2000.

The database was modified in preparation for information retrieval, including the addition of auxiliary fields to facilitate sorting and the exclusion of records not considered relevant or applicable to the BERA. The additional fields included ecosystem classification (i.e., freshwater or saltwater), trophic level (i.e., predator, omnivore, bottom feeder (detritivore), or filter feeder), and taxonomic phylum. The modified database was then queried using several filtering criteria including: tissue type (whole body or egg tissue), ecosystem classification (freshwater species), and BERA COPCs. The query resulted in a dataset of approximately 700 records, which were then screened for duplicate records. Finally, output was further limited to include only survival-, growth-, and reproduction-based endpoints. Data were summarized for the following contaminant groupings: PAHs, pesticides, PCB Aroclors, PCB congeners, TCDD/TCDF, and inorganics (Tables G-2 through G-12 in Appendix G). The majority of the information is for aquatic invertebrate and fish species; however, Tables G-4 and G-10 summarize available effects data for pesticides and TCDD/TCDF in avian egg tissue respectively.

For each contaminant class and general effect category (i.e., mortality, growth, and reproduction), LOAELs and NOAELs were selected for invertebrates and fish species (the selected NOAELs were the highest concentration in which no adverse effect was observed that does not exceed the LOAEL for a given taxon and effect category). Occasionally, either a LOAEL or NOAEL could not be established from the information in the database, resulting in an unbounded estimator of the effect threshold for a given taxon/effect category combination. Table G-1 summarizes the derived NOAELs and LOAELs obtained from this analysis; concentrations are reported as ug/g wet weight. The effect thresholds presented for invertebrate species were compared to the measured COPC concentrations in crayfish for this evaluation.

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3.2.3 Sediment Bioassay

The sediment bioassay study was conceived as part of an integrated, effects-based approach to characterize sediment quality in the study area and provide a holistic evaluation of benthic community health. The laboratory bioassay results were interpreted along with results of sediment chemistry and the macroinvertebrate community study. Study details are described in further detail in the work plan (Harding ESE, 2001a).

The selection of eight sediment sampling stations in the Woonasquatucket River, including background and reference locations in Greystone and Assapumpset, respectively, was based on review of historical sediment chemistry data and consideration of the spatial extent of potential sediment impacts (Harding ESE, 2001). Depositional areas with elevated PAH, PCB, pesticide, and metal concentrations were identified and specific stations were selected where historical sediment chemistry results indicated that various combinations of these potential stressors were present in elevated concentrations.

Three sampling stations were selected in Allendale: APB-SD-4008, APB-SD-4009, and APB-SD-4010 to evaluate sediment quality in the impoundment immediately downriver of the Site (Figure 58). Station APB-SD-4008 is located immediately south of Cap 2 (part of the former tailrace) and the other two stations are just upriver of the Allendale Pond. Elevated concentrations of PAHs, PCBs, pesticides, and metals (e.g., chromium, lead, silver, and zinc) were detected in historical samples collected in the vicinity of these stations. Three sampling stations were also selected in Lyman Mill: LPX-SD-4011, LPX-SD-4012, and LPX-4013; LPX-SD-4011 is situated near the boat launch area adjacent to Lyman Mill Dam and the other sampling stations are located in the upper portion of the pond. In general, historical data suggested that concentrations of PAHs, PCBs, and pesticides were lower than in Allendale; however, concentrations of some metals (e.g., zinc) were similar.

One background location, GMP-SD-5007 (immediately upriver of the Greystone Mill Pond Dam), and one reference location, RAB-SD-5008 (in Assapumpset Brook immediately below the overflow structure from Assapumpset Pond) were sampled. These locations were selected to represent the range of grain size composition, total organic carbon concentrations, and habitats expected at downriver sampling locations in the Woonasquatucket River. Historical data also indicated that the sediments in the vicinity of GMP-SD-5007 contained elevated concentrations of PAHs and metals; these data may be representative of general conditions in the river.

Eight sediment samples were collected and submitted to Great Lakes Environmental Center (GLEC) for whole sediment toxicity testing on two species of macroinvertebrates. Split sediment samples were also submitted for chemical analysis of SVOCs, pesticides, PCBs (Aroclors), metals and miscellaneous parameters including percent moisture, AVS/SEM, grain size, and TOC. Two separate bioassays were performed with sediment

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from each sampling station using the freshwater amphipod *Hyalella azteca* and the freshwater midge *Chironomus tentans*. Sediment samples for toxicity testing were collected between 2 July and 5 July 2001.

The tests and quality control were conducted in accordance with procedures outlined by EPA/600/R-99/064 (USEPA, 1999), the American Society for Testing and Materials standards (ASTM, 2000) and GLEC Standard Operating Procedures (GLEC, 2002). Adverse effects were evaluated by measuring survival, growth, and reproduction through chronic life-cycle exposures in the two test species. In both tests, organisms were exposed to sediment from 6 locations within the site, one upriver background location (Greystone), and one reference location (Assapumpset). Laboratory control sediment was exposed concurrently with the investigation samples. Sixteen replicates were initiated at the start of the experiments.

The *C. tentans* tests were initiated on 1 August 2001 and terminated on 18 September 2001, and the *H. azteca* tests were initiated on 2 August 2001 and terminated on 12 September 2001. No deviation from the work plan was encountered during toxicity and daily monitoring indicated that routine water quality parameters (including ammonia) did not contribute to the observed toxicity.

Six endpoints were measured in the bioassays: 49 day survival, 20 day growth, and reproduction (number of egg cases) were measured in *C. tentans* and 42 day survival and growth and reproduction (average number of young) were measured in *H. azteca*. A two sample t-test was used to compare survival and growth endpoints for the 2 background/reference sites to corresponding survival and growth endpoints for each of the investigation sediment samples for the *H. azteca* tests. In the *C. tentans* test, neither background/reference station met the minimum survival criteria established in the test protocols and the laboratory negative control results were used for the statistical analysis.

3.2.4 Aquatic Invertebrate Community Bioassessment

A benthic macroinvertebrate community study was conducted at nine sampling stations in flowing sections of the Woonasquatucket River and Assapumpset Brook during 3-5 July 2001. The report describing this study is included in Appendix E. The overall objective of the study was to determine whether the benthic macroinvertebrate community in the study area was impaired relative to background and reference sampling locations. As described in the Work Plan (Harding ESE, 2001a), sampling stations were also selected to assess the specific potential point source impacts of the Smithfield WWTP and the discharge of contaminated overburden groundwater into the Woonasquatucket River adjacent to Centredale Manor apartment complex. The FSP (Harding ESE, 2001b) provides a description and rationale for each of the selected sampling locations. To meet these objectives, the FSP (Harding ESE, 2001b) identified a total of ten stations: three stations above the Greystone Mill Pond Dam near the

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Smithfield WWTP, three stations in the upper portion of the Allendale EA; two stations below Allendale Pond; and two stations in Assapumpset Brook.

It was determined during the site reconnaissance that the flow adjacent to, and immediately downstream of, the WWTP was affected by the Greystone Mill Pond Dam and that the selected bioassessment methodology was not appropriate for this section of the river. Two sampling stations were relocated from the vicinity of the WWTP to the reach between the dam and Route 44 (upriver background stations) and a third station was relocated from the vicinity of the WWTP to the river reach adjacent to the Site to allow a more precise delineation of the potential effects of discharging groundwater on the macroinvertebrate community. The downstream station in Assapumpset Brook was eliminated during the site reconnaissance due to a lack of appropriate habitat. Sampling stations are shown in Figure 58.

The benthic bioassessment report, which is included in Appendix E, discusses the methods employed in the benthic community study.

Benthic macroinvertebrates kick samples were collected from nine sampling stations in flowing sections of the river, and ten individual locations were sampled from each of the nine sampling stations during 3-5 July 2001. Samples were collected by placing a 500 µm mesh dip net perpendicular to the substrate and agitating the substrate immediately upriver. At each station, five representative samples were collected from fast and slow water habitats and the individual samples were composited. The physical habitat of the river was also evaluated at each benthic macroinvertebrate sampling station and standard habitat scoring forms were completed. Ten high gradient habitat assessment metrics were evaluated at each station and scored using a numerical scale of 0 to 20 (Barbour et al., 1999). Water quality parameter data including temperature, dissolved oxygen, conductivity, and pH were collected at each station.

Macroinvertebrates were identified in the laboratory to the lowest practicable taxonomic levels and the data were analyzed using seven biological metrics: taxa richness, biotic index, ratio of scrapers to filtering collectors, ratio of Ephemeroptera, Plecoptera, and Trichoptera (EPT) to Chironomidae abundance, percent contribution of the dominant family, EPT richness index, and community loss index (Barbour, 1999). Metric scores for each assessment station were totaled and compared to the total metric scores for the background/reference data (either the Assapumpset station or the mean of the two stations located in the Woonasquatucket River upriver of Route 44).

3.3 RISK CHARACTERIZATION - AQUATIC INVERTEBRATES

This section provides an evaluation of each of the measurement endpoints established to characterize risk to the aquatic invertebrate community.

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3.3.1 Surface Water Benchmarks

Tables 35 through 38 present the comparison of surface water benchmarks to the CT and RME EPCs for Assapumpset, Greystone, Allendale, and Lyman Mill, respectively. Figure 59 summarizes the summed HQs by parameter group for each exposure area. HIs for both the CT and RME EPCs exceed 1 in all EAs, the CT HIs range from 3.1 (Assapumpset) to 290 (Allendale). COPCs contributing most substantially to the HIs are endrin, total PCBs, the TEQ for dioxins/furans (using the fish TEFs), and silver in Allendale Pond. The TEQ for dioxin/furans (using the fish TEFs), DDT, and silver contribute most substantially to the HI for Lyman Mill. In the upriver background area, dissolved metals (particularly cadmium) contribute most substantially to the HI. Ammonia concentrations throughout the study area, including Greystone Mill Pond (immediately downstream of the Smithfield WWTP) do not exceed the chronic ammonia AWQC. No COPC retained for Assapumpset Pond exceeds benchmarks with the exception of the RME concentration for copper (HQ = 1.3). HQs for the inorganic COPCs are fairly consistent throughout the entire Woonasquatucket River with summed HQs ranging from 16 (Greystone) to 37 (Allendale); however, silver was not a risk contributor at the upstream background area and it accounted for the majority of the risk associated with dissolved metals in both Allendale and Lyman Mill.

Table 39 presents the comparison of surface water benchmarks to the CT and RME EPCs for Allendale overburden groundwater. RME and CT HIs are 20,000 and 8,100, respectively. Both RME and CT EPCs for tetrachloroethene and 2,4,6-trichlorophenol exceed respective surface water benchmarks by three orders of magnitude. Figure 60 summarizes the CT- and RME-derived HIs by each exposure area for abiotic media.

3.3.2 Sediment Benchmarks

Tables 40 through 45 present the comparison of sediment benchmarks to the CT and RME EPCs for Assapumpset, Greystone, Allendale, Lyman Mill, Manton and Dyerville, respectively. Figure 61 summarizes the summed HQs by parameter group for each exposure area. HIs for both the CT and RME EPCs exceed 100 in all EAs, the CT HIs range from 110 (Manton) to 4,300 (Allendale and Lyman Mill); CT HIs for Assapumpset and Greystone are 460 and 1,700, respectively. Pesticides and PAHs contribute most substantially to the HIs throughout the study area including the background and reference areas (Figure 61). However, the PAH ESGs (for CT EPCs) range from 0.55 (Greystone) to 2.4 (Dyerville) suggesting that there is much less risk to benthic macroinvertebrates than the results of the individual benchmark comparisons would suggest. The PAH ESGs are within the range of greatest predictive uncertainty regarding benthic effects even if the 11.5-fold adjustment factor (USEPA, 2000a) were to be applied. Figure 62 summarizes the individual PAH ESGs calculated for each sediment sample in the database for which both PAHs and TOC were analyzed. Analytical samples are ranked by the PAH ESG values within exposure area in the figure. The PAH ESGs for

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approximately half of the samples in Allendale exceed 1 and all but one sample do if the adjustment factor is considered (lower bar in Figure 62). In Lyman Mill, the PAH ESG values for 7 of 24 samples exceed one, and as observed in Allendale, with one exception, the adjusted PAH ESGs would exceed one.

EPCs for pesticide compounds, particularly dieldrin, endrin, and chlordane, substantially exceed sediment benchmarks throughout the study area (Figure 61) as do the EPCs for PCB Aroclors. CT estimates for the carbon-normalized SEM indicate that these divalent cations are not likely to cause toxicity to benthos throughout the study area with the exception of Dyerville (CT = 733 $\mu mol/g_{oc}$) under average exposure conditions; however, all RME values exceed the 120 threshold value (USEPA, 2000b) suggesting that localized effects are possible. Figure 63 presents the ranked carbon-normalized excess SEM values for each exposure area (the bar represents the 130 $\mu mol/g_{oc}$ threshold). At only one location (DYP-SD-2065) in Dyerville are divalent cations in sediment pore water likely to be causing toxicity to benthos; the majority of sediments are unlikely to be toxic.

As with the surface water benchmark evaluation, the application of the toxic equivalency approach to invertebrate organisms is highly conservative as there is little evidence that the AhR occurs in these organisms. The TEFs developed for fish that were used to develop TEQs are probably not applicable for this class of endpoint receptors. The TEQs are presented in these tables for the assessment of risk to demersal and pelagic fish, however. Figure 60 summarizes the CT- and RME-derived HIs by each exposure area for abiotic media.

3.3.3 Sediment Bioassay

The laboratory bioassay report is provided in Appendix F. Table 1A in the bioassay report summarizes the overall findings and a comparison of bioassay effects and benchmark screening for each treatment sample is provided in Figure 58.

Hyalella azteca. A significant reduction ($p \leq 0.05$) in survival after 10 days exposure was observed in two samples (APB-SD-4008 and LPX-SD-4012) and in four samples (APB-SD-4008, APB-SD-4009, APB-SD-4010, and LPX-SD-4012) after 28 days and 42 days of exposure when compared to the background/reference control samples (i.e., RAB-SD-5008 and GMP-SD-5007) (Figure 64). Growth, measured as mg average dry weight, was significantly reduced in three samples (APB-SD-4008, LPX-SD-4011, and LPX-SD-4012) after 28 days exposure when compared to the background/reference sediments (Figure 64). There was no significant reduction in growth for any samples when measured at 42 days from test commencement; however, the organisms were transferred to clean water after 28 days following the protocol. Reproduction, measured as the average number of young per female, was similar between investigation and background/reference samples in the majority of the test samples. However, the

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reproduction associated with sample APB-SD-4008 was considerably less than the controls (Figure 65).

Chironomus tentans. The background/reference sediments met the minimum growth criteria, but not the minimum survival criteria for an acceptable control. Specifically, the reference control sediment (RAB-SD-5008) met the minimum requirements for acceptable control survival for day 20 and 49 of the test, while the 10 day survival of 68.7% was below the acceptable level of 70%. The background control sediment (GMP-SD-5007) did not meet the minimum survival criteria for the 10, 20, or 49 day survival endpoints. Consequently, the laboratory control results were used for statistical analysis.

There was a significant ($p \leq 0.05$) reduction in the survival of all treatment samples after 10, 20, and 49 days when compared to the laboratory sediment control (Figure 66). One sample (APB-SD-4010) experienced 100 percent mortality after 10 days exposure. Growth (measured as ash free dry weight) was significantly reduced in one sample (APB-SD-4008) and another sample had 100 percent mortality in all replicates (Figure 66). Reproduction (average number of eggs produced per female) was similar between the laboratory control samples, the background/reference samples, and the remaining treatments. However, the number of egg cases produced was much lower in sample APB-SD-4008 (3 egg cases/female compared to 36 produced by the average control female) (Figure 65). No reproduction occurred in APB-SD-4010 due to the death of all test organisms.

Survival was the more sensitive endpoint for both test species. *H. azteca* survival was significantly ($p \leq 0.05$) reduced in four of the treatment samples (APB-SD-4008, APB-SD-4009, APB-SD-4010, and LPX-SD-4012) and growth was significantly reduced in three of these same four sediment samples (APB-SD-4008, APB-SD-4009, and LPX-SD-4012). Midge survival was significantly reduced in all samples except the reference sample RAB-SD-5008, whereas growth and reproduction of the surviving organisms was reduced in only two of the test sediment samples. *C. tentans* was more sensitive to Woonasquatucket River sediment in these tests because both acute and chronic survival were significantly affected in all treatment samples and in one of the background sediment samples (GMP-SD-5007).

Table 46 presents a summary of the bioassay results and the sediment chemistry for the exposure sediments. Analyte concentrations that exceed sediment benchmarks are shaded in the table. Tables 47 through 49 present correlation matrices for *C. tentans* and *H. azteca* 49-day and 42-day survival and PAH, pesticide/PCBs, and inorganic analyte concentrations, respectively. Strong negative correlations ($r^2 > -0.60$; $n = 11$) between survival and analyte concentrations were identified for the following:

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Species/effect	Analyte	Coefficient
C. tentans 42-d survival	$PAHESG_{TOT}$	-0.95
	Dieldrin	-0.82
	Aroclor-1254	-0.78
	Total Aroclor	-0.78
	Mercury	-0.67
H. azteca 49-d survival	Acenaphthene	-0.75
	Anthracene	-0.60
	Dibenzofuran	-0.66
	Fluorene	-0.67
	Phenanthrene	-0.66
	$PAHESG_{TOT}$	-0.73
	Dieldrin	-0.61
	Mercury	-0.62

It is reasonable to conclude that all, or some combination of these chemical stressors, are contributing to the toxicity observed in the laboratory bioassays; however, the study design is not adequate to determine causality. Figure 58 displays the significant biological effects observed in the laboratory and ratios of detected sediment analytical results and sediment benchmarks for the above set of analytes.

3.3.4 Critical Body Residues

Measured analytical chemistry of aquatic biota tissue was compared to available CBRs for invertebrate species. The CT EPCs for each tissue type and exposure area were compared to CBR NOAELs and LOAELs as summarized in Table G-1.

Crayfish Tissue. Tables 50 through 53 present the calculated HQs for COPCs detected in crayfish tissue. Figure 67 summarizes these results graphically by EA. The TEFs for fish were used to derive a TCDD TEQ for comparison to the invertebrate derived CBRs. Both NOAEL- and LOAEL-based HIs are very consistent across the EAs; the LOAEL-derived HIs range from 63 (Allendale) to 100 (Assapumpset), the NOAEL-derived HIs range from 13 (Assapumpset) to 19 (Lyman Mill). For these comparisons, the LOAEL-derived HIs were often larger than the NOAEL-derived HIs because the CBR effect range for some analytes was not lower bounded.

Only the concentrations of several metals (including barium, copper, manganese, and zinc) detected in crayfish tissue consistently exceed available CBRs for invertebrates.

Emerging Insects/Swallow Stomach Content Samples. Tables 54 and 55 present a comparison of the TCDD TEQ (based on fish TEFs) for emerging insect tissue for Greystone and Lyman Mill, respectively, to the invertebrate CBR. Tables 56 through 59

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present a similar comparison for stomach content samples collected from tree swallow nestlings collected from Greystone, Allendale, Lyman Mill, and Manton, respectively. Figure 68 displays the results of these comparisons for each exposure area, and indicates that the measured dioxin and furan congeners concentrations in emerging insects collected from the Site are not predicted to pose a risk to these organisms.

Figure 69 summarizes the NOAEL- and LOAEL-derived HIs by each exposure area for the comparison of CBRs to aquatic invertebrate tissue.

3.3.5 Benthic Macroinvertebrate Community Analysis

Overall, benthic macroinvertebrate sampling stations in the Woonasquatucket River had good to excellent habitats for supporting benthic communities. Taxa richness was high at all stations with none of the stations having numerical dominance by a single taxon greater then 27.2%, and most being below 20%. EPT to Chironomidae abundance ratio and EPT richness metrics received an optimal score of 6 at all stations except one (GMP1 and GMP2, respectively [both in Greystone]). Analysis of benthic data from six of the eight experimental stations indicated no impairment and two stations (GMP1 and APB4) indicated non to slight impairment when compared to the upgradient background station. In summary, benthic macroinvertebrate communities in the Woonasquatucket River, downstream of the source area, do not appear to be substantially impaired (Appendix E).

The RIDEM conducted rapid bioassessment screening every year from 1992 though 1998. Forty freshwater sites within the state of Rhode Island were initially selected for sampling and increased to 45 locations the following year. All selected freshwater sites were representative of the different watershed environments within the state and included different water quality types (Gould, 1998). Sampling was conducted with a hand-drift net and a minimum of 100 organisms were collected (Gould, 1996). The Woonasquatucket River was sampled each year at the Eagle Street Bridge in Providence and in this lower reach of the river; RIDEM found limited habitat for macroinvertebrates due to embedded sediment structure, channelization, and obvious effects of urbanization from both point and non-point sources (Gould, 1993; 1994; 1995; 1996; 1997; and 1998).

The bioassessment findings for the Woonasquatucket River station generally concluded that the benthic community was moderately impaired relative to the reference stations (located on the Wood River all years except for 1992, when a station in the Fall River was used). However, during the summer of 1992 and 1995, the river was classified as being severely impaired, and abundance and diversity were generally low with the fauna dominated by oligochaetes, mollusks, and hydropsychid caddisflies. In some years only a few invertebrates were collected (Gould, 1993; 1994; 1995; 1996; 1997; and 1998). Although these data were collected several miles downstream of the study reach, they provide a nearly decade long picture of the composition and temporal variability of the benthic community in the lower portion of the Woonasquatucket River.

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3.3.6 Emergent Insect Community Assessment

The damselfly *Enallagma* sp. was the numerically dominant taxon, comprising approximately 64% of the insects collected in Greystone (Appendix E). Approximately 26 percent of the organisms collected were non-biting midges (family Chironomidae), with other taxa comprising 9% of the total insects collected. *Enallagma* sp. was also the numerically dominant taxon in the Lyman Mill samples and comprised approximately 89% of the insects collected. Chironomids comprised 9% of the organisms collected and other taxa individually accounted for less than 2% of the total number of organisms collected. A total of 12.8 and 7.5 grams of aquatic insects were collected from the pooled traps in Greystone Mill and Lyman Mill, respectively.

No significant difference in the biomass samples of emergent insects and no apparent difference in the taxonomic composition of emergent aquatic insect community was noted between Greystone Mill and Lyman Mill (Appendix E).

3.3.7 Risk Uncertainties

General risk uncertainties affecting the BERA are summarized and discussed in Section 10. Specific uncertainties relevant to this assessment endpoint include the following:

- The surface water/sediment benchmarks employed in the assessment were not site-derived values (although surface water benchmarks for certain inorganic COPCs were hardness-adjusted to reflect Site conditions) and were generally selected to be conservative estimators of effect.
- The surface water and sediment analytical results used to develop the EPCs are likely to be generally representative of the exposure conditions experienced by aquatic receptors. However, relatively few surface water and sediment results were available for the lower EAs evaluated in the BERA. In addition, the effects of the Allendale Dam breach in April 2001 on the representativeness of the sediment analytical data for Lyman Mill is uncertain although it is anticipated that there was a short-term increase in the bioavailability of potentially bioaccumulative compounds.
- Although elevated VOC concentrations were detected in both overburden groundwater at the Site and in vapor diffusion samplers placed in the adjacent reach of the Woonasquatucket River, no VOC was selected as a COPC. This lack of correspondence between the data sets suggests that the available surface water data may not be representative of potential VOC exposures in the lotic portion of the Allendale EA. This is likely accounted for by a lack of co-location between surface water samples and portions of the reach where contaminated overburden groundwater is discharging. However, these exposures were the specific focus of the macroinvertebrate community study.
- Relatively few CBRs are available for the COPCs identified in aquatic invertebrate tissue (including emerging insect, swallow nestling stomach contents, and crayfish).

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Risks for those COPCs lacking CBRs were likely under-estimated. In addition, CBRs for other taxa were used when invertebrate data were lacking and the extent to which effects can be extrapolated between different types of receptors is unknown. With respect to exposure to dioxin, furan, and co-planar PCBs, the BERA assumed that invertebrates would be as sensitive as fish receptors. This certainly resulted in risks being over-estimated because it is unlikely that the AhR toxicity model applies to invertebrates.

- The assessment of emerging insect productivity was affected as no information could be obtained for the Allendale EA due to the low water levels in 2001 following the breach of the Allendale Dam. In addition, the relatively low production in both Greystone and Lyman Mill during the field programs necessitated the compositing of the sample results and information on the spatial variability within EAs was lost.
- Unacceptably high mortality was obtained in the midge sediment bioassay for both background and reference stations and the laboratory negative control results were used in the statistical analysis. Moreover, the specific chemical stressors responsible for the observed toxicity in some of the bioassays were not definitively identified. Consequently, it is uncertain whether the positive response for this measurement endpoint is due to Site-related COPCs or not.

3.4 RISK ASSESSMENT SUMMARY – AQUATIC INVERTEBRATES

Tables 60 and 61 present the incremental risk evaluation for the surface water medium based on comparison of each exposure area to Greystone Mill and Assapumpset, respectively. Incremental risk estimates were derived by subtracting the risk estimate derived for either the background (i.e., Greystone) or reference (i.e., Assapumpset) area from the corresponding estimate at an EA. The range of Incremental risks was estimated using risk estimates based on either NOAELs/LOAELs (in the case of comparison of CBR estimates with biological tissue data) or RME/CT exposures (in the case of benchmarks comparisons with COPCs in abiotic media). Incremental risks were calculated only for those COPCs whose NOAEL- or RME-based HQs are greater than 1, and which in combination with other selected analytes contribute at least 95% to the total risks. In addition, only COPC, whose Site risk estimates exceeded those derived at the background or reference area, were considered in the incremental risk evaluation.

Primary contributors to the incremental risk for aquatic invertebrates include: TCDD TEQ, PCB Aroclors, and several pesticides. As discussed above, application of the TEF approach for assessing potential impacts to invertebrate species is highly questionable given the general absence of the AhR in this taxonomic group. Of the remaining COPCs, only Aroclor 1254 appears to clearly be related to the Centredale source area.

Tables 62 and 63 present a similar comparison of the findings at the background and reference areas to EAs for sediment; and Tables 64 and 65 present incremental risk estimates for the crayfish tissue/CBR comparison. In sediment, the primary contributors

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to the incremental risk include TCDD TEQ, PAHs, and PCB Aroclors; the latter two categories of COPCs were both identified as potential stressors in the laboratory toxicity tests. Crayfish tissue residues measured in the study area appear relatively consistent with similar data collected in Greystone and Assapumpset (Tables 65 and 66).

Figure 70 presents a summary of the incremental risk evaluation for aquatic invertebrates.

The weight-of-evidence integration for the measurement endpoints evaluated for the aquatic invertebrate community assessment is summarized in Table 66. As part of the integration of the various lines of evidence, it may be helpful to consider the information presented in Table 66 as a surface with a fulcrum located under the page in the center of the matrix (MWOEW, 1995). A conclusion of substantial risk of harm can be obtained if it is determined that the top of this imaginary surface would likely tip "down" (i.e., the preponderance of evidence favors a positive finding of substantial risk to the endpoint receptor; a negative finding can be obtained if it is concluded that the top of the surface is likely to tip "up".

As discussed in Section 2.6.1.1, this assessment included a risk evaluation for two separate aquatic habitats that occur within the study area: lotic (particularly in the vicinity of the Site) and lentic (depositional areas typically associated with the various impoundments along the Woonasquatucket River. While the majority of the identified measurement endpoints are applicable to both aquatic habitats, the sediment bioassay (endpoint C) and macroinvertebrate community study (endpoint E_1) were considered separately in the evaluation of lentic and lotic habitats, respectively.

Lentic Habitat. The comparison of COPC concentrations detected in surface water and sediment to applicable criteria and guidelines (measurement endpoints A and B in Table 66) provide a strong indication of harm but were assigned only a "low-medium" weight in the *a priori* scoring (Table 15). The sediment bioassay (measurement endpoint C) also provides a strong indication of harm and was assigned a "medium-high" weight. The comparison of aquatic invertebrate tissue concentrations to CBRs provides a weaker indication of effects. Crayfish tissue concentrations exceed the CBRs for several inorganic analytes but concentrations detected in emerging insects and swallow stomach contents do not.

The measurement endpoint related most directly to the community level assessment endpoint in lentic habitats (E_2 in Table 66) does not suggest that the aquatic invertebrate community within the study area has been harmed although the limited data are not particularly compelling. With the exception of this endpoint, however, the answer to all risk questions evaluated for lentic habitats is in the affirmative, and it is concluded that aquatic macroinvertebrate community is at substantial risk of harm.

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Lotic Habitat. The assessment of risks to aquatic macroinvertebrate communities associated with riffle/run habitat, particularly in the vicinity of the Site is based on consideration of measurement endpoints A, B, D, and E_1 (Table 66). The negative finding for the aquatic macroinvertebrate community study is sufficient to counterbalance the other endpoints, especially when it is appreciated that the sediment EPCs are biased high in depositional samples relative to those obtained in riffle/run areas.

Integrating these results, it is concluded that the aquatic macroinvertebrate community associated lotic environments with the study area is not at substantial risk of harm due to the presence of contaminant release (including VOCs) from the Centredale Manor source area. However, the aquatic invertebrate community associated with the more depositional and low-gradient areas within the study area appear to be at substantial risk of harm with individual populations likely to experience reduced survival, growth, and/or reproductive success (as indicated in the laboratory bioassay study).

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4.0 FLOODPLAIN INVERTEBRATE COMMUNITY RISK EVALUATION

This section evaluates risks to the floodplain invertebrate community using the measurement endpoints discussed in Section 2. The evaluation of the aquatic invertebrate community is presented in Section 3. The measurement endpoints, sampling strategy, and methods are described, followed by an evaluation of floodplain soil and biological tissue. Benchmark concentrations from the literature are discussed. Measured effects on floodplain soil invertebrates are evaluated based on the results of the floodplain soil community bioassessment study and comparison of soil chemistry to soil invertebrate benchmarks. The results from the analytical chemistry and community analyses are then integrated to characterize risks to this community.

Further details of the study design rationale and methods are presented in the FSP (Harding ESE, 2001a; 2001b).

4.1 EXPOSURE ASSESSMENT - FLOODPLAIN INVERTEBRATES

In this section, COPC exposure by floodplain invertebrates is assessed using both historical and additional analytical data collected in 2001. Table A-1 summarizes the information on the analytical samples for surface soil and earthworm tissue that were used in this evaluation. These data were collected throughout the study area to represent a range of exposure to different contaminants and to integrate abiotic media concentrations with toxicity and macroinvertebrate community structure measured in co-located samples. In addition, analytical chemistry data for co-located biota tissue and associated soil samples were used to develop BSAFs for predictive modeling and development of PRGs.

All chemistry results as reported by the laboratory, including data qualifiers are presented in the DER (Harding ESE, 2003). The laboratory data were evaluated in terms of completeness, holding times, instrument performance, accuracy, precision, method reporting limits, and field quality control samples. During the quality assurance review, no data were rejected and all data were considered useable as reported.

4.1.1 Floodplain Soil Quality Evaluation

Analytical chemistry data for floodplain soils were statistically summarized and compared to benchmark concentrations protective of soil invertebrates. Tables 67 and 68 present the floodplain soil EPC concentrations for the CT and RME, respectively. EPCs were developed for Greystone, Allendale, and Lyman Mill.

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4.1.2 Soil Invertebrate Community

A soil invertebrate community study was conducted at eleven sampling stations in floodplain soils associated with the Woonasquatucket River during 16 and 29 July 2001. The report describing this study is included in Appendix E. The overall objective of the study was to determine whether the soil macroinvertebrate community in the study area was impaired relative to upstream sampling locations (Harding ESE, 2001b). The FSP (Harding ESE, 2001b) provides a description and rationale for each of the selected sampling locations. Three sampling stations (RWR-FP-5001, RWR-FP-5002, RWR-FP-5003) were established in the upriver background area in the Woonasquatucket River floodplain located below Greystone Mill Pond; three sampling stations were established in Allendale (CMS-FP-4001, CMS-FP-4002, and CMS-FP-4003); three sampling stations were located in Lyman Mill: one station behind the Boys and Girls Club (LPX-FP-4004) and two stations along the eastern shore of Lyman Mill Pond (LPX-FP-4006 and LPX-FP-4007). The FSP also identified an fourth sample station in Lyman Mill and at a reference station in Assapumpset; however it was determined during the site reconnaissance that suitable habitat was unavailable to establish these two stations as indicated in the sampling plan. These two stations were relocated to Allendale (CMS-FP-4005) and the upstream background area (RWR-FP-5004). Sampling stations are shown in Figure 58.

At each sampling station, vegetation, and soil types were characterized after which surface litter and debris were removed in preparation for sample collection. A 10 foot by 10 foot area was delineated with nylon twine and a sampling grid consisting of 1 square foot cells was established (Appendix E). A decontaminated shovel was used to sample the soil from 5 randomly selected cells within each grid. Each replicate sample was sorted for 5 minutes and all soil invertebrates were removed, rinsed with distilled water, placed in a labeled sample container, and preserved with 10% formalin. The soil invertebrate community samples were returned to the Normandeau laboratory in Bedford, New Hampshire for taxonomic identification and enumeration.

Following the initial collection of soil invertebrate community samples, additional soil invertebrates within the sampling grid were sampled by picking from the soil directly or by applying the probes of a backpack electroshocker to force the organisms to the surface. After collection of the required tissue mass for tissue analysis, earthworms were rinsed in distilled water, placed in sample containers and shipped under Chain-of-Custody to the analytical laboratory. The study report is included in Appendix E.

4.1.3 Earthworm Tissue

Eleven composite earthworm samples were collected at each of the sampling stations established for the soil invertebrate community analysis. Earthworm tissue samples were collected in accordance with the SOP (Harding ESE, 2001b) and submitted for chemical

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analysis to determine whether tissue concentrations pose a risk of direct harm to invertebrates or to insectivorous wildlife that forage on these organisms. Composite floodplain soil samples were also collected at each of the earthworm sampling stations in order to derive BSAFs for predictive modeling of contaminant biouptake.

Sampling stations were selected based on a review of the historical sediment data in order to provide a range of potential exposures to dioxins and PCBs (Harding ESE, 2001b). Four sampling stations were established in Allendale (CMS-EW-4001, CMS-EW-4002, CMS-EW-4003 and CMS-EW-4005); three sampling stations were located in Lyman Mill: one station behind the Boys and Girls Club (LPX-EW-4004) and two stations along the eastern shore of Lyman Mill Pond (LPX-EW-4006 and LPX-EW-4007) (Figure 58).

A total of four background earthworm samples were collected: RWR-FP-5001, RWR-FP-5002, RWR-FP-5003, and RWR-EW-5004 in the reach of the Woonasquatucket River between Greystone Mill Pond and Route 44. These sample stations were located in comparable earthworm habitat and tissue levels in the samples collected upriver of the site were used to establish background levels of the analyzed compounds.

Tables 69 and 70 present the earthworm EPC concentrations for the CT and RME, respectively. EPCs were developed for the Greystone, Allendale, and Lyman Mill.

4.2 EFFECTS ASSESSMENT - FLOODPLAIN INVERTEBRATES

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

4.2.1 Floodplain Soil Benchmarks

TRVs used to evaluate the potential hazard associated with exposure to the surface soil EPCs are documented in Table D-3.

4.2.2 Critical Body Residues

Section 3.2.2 describes the development of the CBRs that were used in the BERA. The NOAEL and LOAEL values for invertebrate organisms presented in Table G-1 were compared to the earthworm tissue residue data.

4.2.3 Floodplain Invertebrate Community Survey

Community survey results are discussed in Section 4.3 and details of the study are presented in Appendix E.

4.3 RISK CHARACTERIZATION - FLOODPLAIN INVERTEBRATES

This section provides an evaluation of each of the measurement endpoints established to characterize risk to the floodplain soil invertebrate community.

4.3.1 Floodplain Soil Benchmarks

Tables 71 through 73 present the comparison of soil invertebrate benchmarks to the CT and RME EPCs for Greystone, Allendale, and Lyman Mill, respectively. Figure 71 summarizes the summed HQs by parameter group for each exposure area. The CT HIs range from 170 (Allendale) to 230 (Lyman Mill); CT HIs for Greystone are 850. Several PAH compounds, pesticides, and chromium contribute most substantially to the HIs throughout the study area including Greystone (Tables 71 through 73).

4.3.2 Critical Body Residues

Measured analytical chemistry of earthworm tissue was compared to available CBRs for invertebrate species. The CT EPCs for each exposure area were compared to CBR NOAELs and LOAELs as summarized in Table G-1.

Table 74 through 76 present the calculated HQs for COPCs detected in earthworm tissue. Figure 72 summarizes these results graphically by exposure area. The TEFs for fish were used to derive a TCDD TEQ for comparison to the invertebrate derived CBRs. Both NOAEL- and LOAEL-based HIs are very consistent across the EAs; the LOAEL-derived HIs range from 460 (Lyman Mill) to 710 (Allendale), the NOAEL-derived HIs range from 2,400 to 4,400 for these same two EAs, respectively. CT EPCs for several inorganic analytes including aluminum, barium, cadmium, copper, lead, nickel, selenium, and zinc exceed CBRs in each exposure area including Greystone. CBRs for pesticides (with the exception of chlordane), Aroclor mixtures, and TCDD TEQs (dioxin/furan and PCBs) are not exceeded in any area.

4.3.3 Floodplain Invertebrate Community Survey

A total of 19 invertebrate taxa were collected from the 11 sampling stations. Seven species of earthworms, six from the family Lumbricidae and one representative of the family Megascolecidae, were collected. Earthworms numerically dominated the taxonomic community samples representing approximately 73%, of the overall fauna collected (Appendix E). *Aporrectoda rosea* dominated the upstream background stations and *Lumbricus rubellus* was most dominant in the experimental stations. Other invertebrates collected include arthropods, gastropods, and nematodes.

The floodplain soil invertebrate communities along the Woonasquatucket River appear to be comprised of organisms that are typically found in wetland and forested habitats along stream banks (Appendix E). The number of taxa found at stations downstream of the Site

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was generally greater than found in the background area (Appendix E). Organisms found at upstream background stations were also found at downstream stations and no organisms were restricted to upstream stations exclusively. Based on a comparison of the relative abundance of organisms, the species diversity, and the overlap of shared fauna, Normandeau Associates concluded that no adverse effects were evident in the floodplain invertebrate communities associated with Allendale Pond and Lyman Mill Pond relative to the upriver background locations.

4.3.4 Risk Uncertainties

General risk uncertainties affecting the BERA are summarized and discussed in Section 10. Specific uncertainties relevant to this assessment endpoint include the following:

- The surface soil benchmarks employed in the assessment were not site-derived values and were generally selected to be conservative estimators of effect.
- Relatively few CBRs are available for the COPCs identified in floodplain invertebrate tissue (e.g., earthworm). Risks for those COPCs lacking CBRs were likely under-estimated. In addition, CBRs for other taxa were used when invertebrate data were lacking and the extent to which effects can be extrapolated between different types of receptors is unknown. With respect to exposure to dioxin, furan, and co-planar PCBs, the BERA assumed that invertebrates would be as sensitive as fish receptors. This certainly resulted in risks being over-estimated because it is unlikely that the AhR toxicity model applies to invertebrates.

4.4 Risk Assessment Summary – Floodplain Invertebrates

Tables 77 and 78 present the incremental risk evaluation for the floodplain soil/benchmark and earthworm tissue/CBR comparisons conducted in Allendale and Lyman Mill as compared to the upriver background area (Greystone). For the soil/benchmark endpoint, incremental risks for the CT EPCs in Allendale and Lyman Mill range from 0.53 to 8.7 (endrin in Allendale). Incremental risks for several pesticides (including 4,4'-DDT, 4,4'-DDE, dieldrin [Allendale only], and endrin [Allendale only]) and zinc (Lyman Mill only) exceed one (Table 77). The incremental risks based on RME EPCs range from 0.62 to 49 (4,4'-DDT in Allendale), with estimates for 4,4'-DDE, dieldrin, and endrin in Allendale all greater than 10.

For the earthworm/CBR endpoint, incremental risks for the LOAEL CBRs in Allendale and Lyman Mill range from 0.081 to 200 (Table 78). Incremental risks based on LOAEL CBRs for several inorganics (including aluminum, barium, lead, vanadium, zinc) and chlordane (Allendale only) exceed one (Table 78). Incremental risks based on NOAEL CBRs range from 6.9 to 1,600 (aluminum in Allendale). Incremental risks for these two measurement endpoints are graphically depicted in Figure 70. As discussed in Section 5.5, the soil invertebrate community endpoint explicitly considers the incremental

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component of risk associated with the Study Area because the conclusions are relative to the Greystone background area.

The weight-of-evidence integration for the measurement endpoints evaluated for the aquatic invertebrate community assessment is summarized in Table 79. The comparison of COPC concentrations detected in floodplain soil and earthworm tissue to applicable surface soil benchmarks and CBRs (endpoints A and B in Table 79) provide a strong indication of harm but were assigned only a "low-medium" weight in the *a priori* scoring (Table 15). The measurement endpoint related most directly to the community level assessment endpoint (endpoint C) does not suggest that the soil invertebrate community within the study area is at substantial risk of harm. While the first two measurement endpoints are answered in the affirmative, (noting the significant uncertainties associated with each of these which would tend to result in potential effects being over-estimated), the available soil invertebrate community data supports a conclusion that Site floodplain areas have a level of ecological integrity that is consistent with background conditions.

Integrating these results, it is concluded that the floodplain soil invertebrate community associated with the study area is not at substantial risk of harm due to the presence of contaminant release from the Centredale Manor source area. Localized invertebrate populations may experience reduced survival, growth, or reproductive success (as suggested by the soil and CBR benchmark comparison endpoints) but these effects do not appear to extrapolate to the community level.

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5.0 DEMERSAL FISH POPULATIONS RISK EVALUATION

This section evaluates risks to demersal fish populations using the measurement endpoints discussed in Section 2. The measurement endpoints, sampling strategy, and methods are described, followed by an evaluation of the exposure of surface water, sediment, and biological tissue. Measured effects on demersal fish are evaluated based on the results of fish community bioassessment and ichthyoplankton studies, a laboratory ELS bioassay, and comparison of measured fish tissue concentrations to CBRs. The results from the analytical chemistry and community analyses are then integrated to characterize risks to this community.

The measurement endpoints are discussed in this section. Further details of study design rationale and methods are discussed in the FSP (Harding ESE, 2001a; 2001b).

5.1 EXPOSURE ASSESSMENT - DEMERSAL FISH

As described in Section 2.2.3.4, demersal fish feed predominantly on benthic invertebrates and consequently have a substantial direct exposure to sediments. Therefore, they represent a unique exposure pathway that is important to consider in assessing risks associated with contaminated sediment.

In this section, COPC exposure by demersal fish is assessed using both historical and additional analytical data collected in 2001. Table A-1 summarizes the information on the analytical samples for fish tissue that were used in this evaluation. These data were collected throughout the study area to represent a range of potential exposures. In addition, analytical chemistry data for biota tissue and associated sediment samples were used to develop BSAFs for predictive modeling and development of PRGs.

All chemistry results as reported by the laboratory, including data qualifiers are presented in the DER (Harding ESE, 2003). The laboratory data were evaluated in terms of completeness, holding times, instrument performance, accuracy, precision, method reporting limits, and field quality control samples. During the quality assurance review, no data were rejected and all data were considered useable as reported.

5.1.1 Fish Tissue

Contaminant concentration data for fish tissue provide a direct measure of exposure of fish as well as of wildlife receptors that consume fish as part of their diet. Three types of biota were targeted for collection based on the combined needs of the BHHRA and BERA; sampling of fish species that represent different trophic levels within the fish community is important as the effects of trophic status are known to influence exposure and uptake potential. The FSP indicated that the three targeted species would include the

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American eel, a turtle species, and a finfish (e.g., largemouth bass, bluegill sunfish) (Harding ESE, 2002a).

Four species of fish were collected contemporaneously with the fish community study for chemical analysis of whole body or fillet/offal tissues. Concentration data for both fillet and offal tissue were obtained for largemouth bass so that these samples could be used for both the BERA and BHHRA (which evaluated human exposure to the fillet or edible portion of the fish sample only). To estimate whole body fish concentrations (necessary for the BERA), the concentration data for the separate tissues were combined on a per weight basis. Species selection was determined following initial survey collections that identified the most abundant and widely distributed species throughout the study area. The white sucker was selected for tissue sample collection; representing the demersal fish trophic category, because is was common in most of the targeted sampling areas. The brown bullhead was selected as an alternative species to be sampled in areas where the white sucker was absent. Fish were caught using the same sampling procedures used in the fish community bioassessment study. Table A-1 (Appendix A) summarizes the numbers and types of demersal fish samples collected during the field program.

Tables 80 and 81 present the fish EPC concentrations based on white sucker or brown bullhead whole body tissue analytical data for the CT and RME, respectively. EPCs were developed for Assapumpset, Greystone, Allendale, and Lyman Mill. No white sucker sample was collected in Assapumpset and the brown bullhead was used as a surrogate demersal species for estimating EPCs in this exposure area. Tables 82 and 83 present the fish EPCs for American eel whole body tissue analytical data for the CT and RME, respectively. EPCs for American eel were developed for all EAs except Manton.

5.2 EFFECTS ASSESSMENT - DEMERSAL FISH

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

5.2.1 Demersal Fish Population Assessment

A fish population and community study was completed in both stream and lacustrine habitats within the study areas in order to assess the general health of the fish assemblage. Index of Biotic Integrity (IBI) samples were collected from lacustrine habitat associated with Assapumpset, Greystone, and Lyman Mill. Stream samples were collected from three stations in the Woonasquatucket River: one below Greystone Mill Pond, one in Allendale, and one below the Allendale Pond dam. Taxonomic and meristic data were collected for all fish collected during the study and used to classify the general condition of the fish community in each experimental station relative to the background stations (USEPA, 1999). The FSP (Harding ESE, 200b) provides details concerning the sample design and objectives of this study.

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Impoundment samples were collected in five sampling areas, two experimental stations in Lyman Mill (LPX1, LPX2), two background stations in Greystone (GMP1, GMP2), and one reference station in Assapumpset (APB1). The breaching of the Allendale Pond dam precluded collecting samples from this EA. In each sampling area, fish were collected along the shoreline from a boat traveling at idle speed for a duration of 10 minutes. Additionally, IBI samples were collected from three stream locations on the Woonasquatucket River; below Greystone Mill Dam (RWR1), in Allendale in the vicinity of the tailrace confluence (APX1), and in Lyman Mill immediately downstream of the Allendale Pond Dam (LPX3). Fish were collected using backpack electroshockers and nets.

All captured fish were placed in a live well, identified to species, weighed to the nearest gram, and total length measured to the nearest millimeter. During fish processing, observations of deformities, lesions, and tumors were recorded in a field notebook. Representative samples were photo documented and scale samples from representative specimens of each species were removed for later age determination. All fish not retained for tissue analysis were then released.

Following the IBI protocol (Barbour et al., 1999), IBI metrics from experimental stations were compared to background stations, and individual metric scores were summed to determine total IBI scores for each sampling station. The following IBI metrics (each of which is defined in the study report) were assessed:

- Total number of species
- Number of benthic insectivores
- Number of water column species
- Number of sucker species
- Number of intolerant species
- Percent white sucker
- Percent omnivores
- Percent insectivores
- Percent top carnivores
- Density of individuals
- Percent hybrids
- Percent diseased individuals

Finally, the station's integrity class was determined by comparing the station's total score with IBI protocol criteria. A detailed description of metrics to assess species richness and composition, tropic composition, and fish abundance and condition is presented in the fish community study report, which is included in Appendix E.

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5.2.2 Ichthyoplankton Study

As a companion study to the fish IBI, a larval fish (ichthyoplankton) study was conducted to evaluate potential differences in population recruitment between EAs within the Study Area and background areas. As indicated in Section 2.4.1.2, fish embryos and larvae are known to be sensitive to many of the COPCs evaluated in the BERA and a field measurement of the abundance and diversity of this community provides a direct measure of effect that has potential population-level implications for this receptor category.

Ichthyoplankton sampling stations were established in Assapumpset, Greystone, Allendale, and Lyman Mill. The FSP (Harding ESE, 200b) provides details concerning the sample design and objectives of this study. The study report is included in Appendix E.

Ichthyoplankton community bioassessments were conducted in three separate ponds in the vicinity of the Site to obtain community measures of species richness and relative abundance. Larval fish sampling was conducted in May and June of 2001 and consisted of collecting three replicate surface tows at four discrete locations on Lyman Mill Pond, three discrete locations in Greystone Mill Pond, and two discrete locations on Assapumpset Pond. Due to sampling difficulties during the May sampling round, associated with extremely abundant aquatic macrophytes clogging the sampling net, a second sampling round was not carried out on Assapumpset Pond in June. Additional qualitative dipnet samples were also collected from two locations in Allendale tailrace during both sampling events due to the shallow water depth of the pond which precluded use of a boat and plankton tow.

Ichthyoplankton samples were individually placed in a labeled sample container and preserved with 10% formalin. The samples were returned to Normandeau's biological laboratory in Bedford, New Hampshire for sorting, taxonomy, and enumeration of species. Details of the sampling and analysis conducted are presented in FSP (Harding ESE, 2001b) and in the study report, which is included in Appendix E.

5.2.3 Critical Body Residues

Section 3.2.2 describes the development of the CBRs that were used in the BERA. The NOAEL and LOAEL values for fish species presented in Table G-1 were compared to the white sucker and brown bullhead (Assapumpset only) whole body tissue residue data.

5.2.4 Partial Life Cycle Bioassay

A laboratory bioassay was conducted to evaluate the lethal and sublethal effects of TCDD, select PCB congeners, and HCX on fish embryos and larvae. This study was intended to provide corroborating information for the ichthyoplankton survey and both

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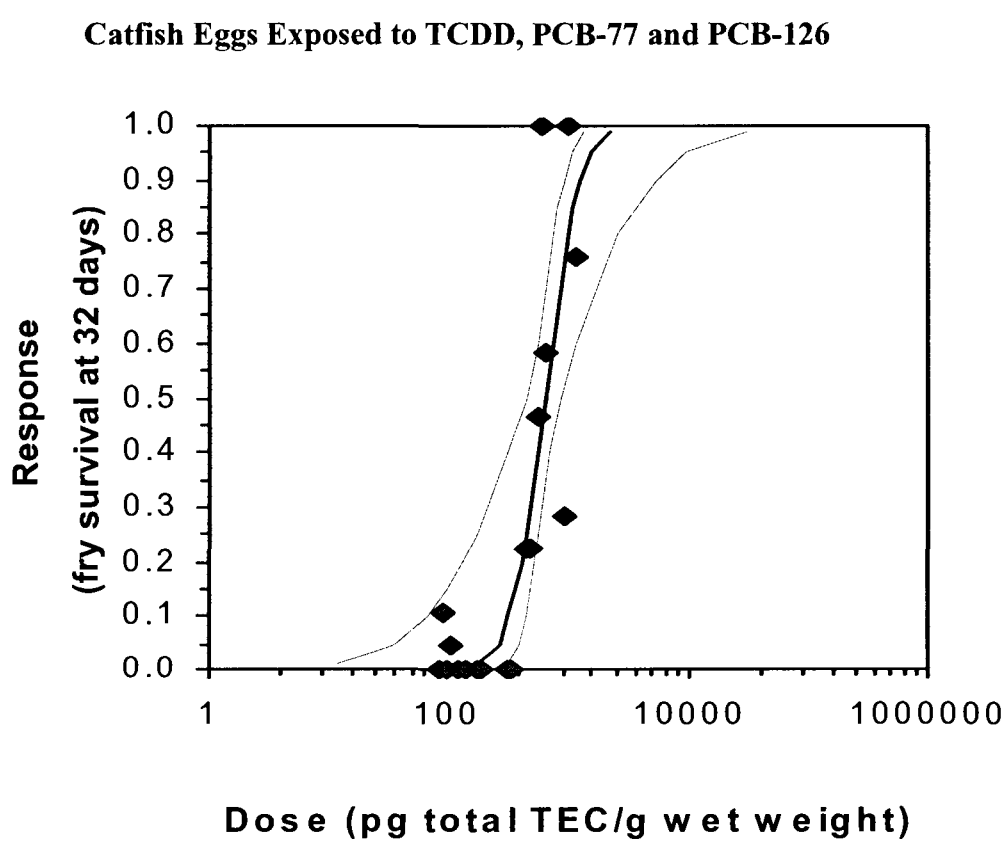
are focused on the most sensitive fish life stage and COPCs believed to be related to the Site. The dose-response data generated from this study can be related to sediment contaminant levels through development of maternal fish/egg transfer factors and site-specific BSAFs.

The study design and methodology employed is described in the Draft Final ELS report (Appendix H). The study involved exposure of fertilized channel catfish (*Ictalurus punctatus*) eggs to a dilution series of a chemical mixture synthesized to emulate the COPC contents of whole fish collected from Allendale Pond. The catfish species was selected because it is an omnivorous species that forages on the river bottom and thus is directly exposed to sediment-borne contamination. Eggs are deposited on the river bottom and embryos could be exposed through direct contact with contaminated sediment as well as the dose received maternally.

Both range-finding and definitive tests were conducted. The latter test included two chemical mixtures: a dilution series of dioxin/PCB and a dilution series of dioxin/PCB/HCX. The tests were conducted for 32 days and consisted of a total of 14 treatments (control, solvent control, 12 test concentrations [six of each mixture]). There were three replicates per treatment with an additional nine replicates for each of the two mixtures to supply tissue for chemical analysis. Each replicate container was initiated with 20 eggs which were randomly reduced to 10 larvae post hatch. Chemical exposure occurred from Day 0 to Day 2. Study endpoints included time to hatch, percent hatch, post hatch survival, observable abnormalities, growth, and body burden in eggs just prior to hatching (i.e., Day 6).

The following figure presents the dose-response results obtained from the TCDD, PCB-77 and PCB-126 exposure treatment.

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The Effective or Inhibition Concentrations (EC/IC) for 10 and 25% of the exposed population (effect/inhibition concentration 10 (EC/IC_{10}) and effect/inhibition concentration 25 (EC/IC_{25}), respectively) threshold range derived from the ELS bioassay (Appendix H) were used to assess the body burdens of individual fish with respect to potential ELS effects to fish eggs. Lipid-normalized TEQs (for dioxin/furan and PCB) were estimated based on the congener concentrations detected in individual fish samples and measured fish lipid content. The TEQs were then graphically compared directly to the threshold ranges discussed in Section 6.4.6 assuming no diminution of concentrations between maternal and egg tissue on a lipid basis.

5.3 RISK CHARACTERIZATION - DEMERSAL FISH

5.3.1 Fish IBI Study

A total of 12 species of fish were collected during the fish community bioassessment study with representatives of the following trophic categories: omnivore (white sucker, golden shiner), insectivore (pumpkinseed [*Lepomis gibbosus*]/redbreast sunfish [*L. auritus*], bluegill, spottail shiner (*Notropis hudsonius*), brown bullhead) and piscivores

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(black/white crappie [*Pomoxis nigromaculatus* and *P. annularis*], American eel, chain pickerel, largemouth bass, and rainbow trout). With the exception of the rainbow trout (collected in the upper section of Greystone and likely a stocked fish) all species are classified as being either tolerant or moderately tolerant to the effects of pollution.

Of the species collected, the Brown Bullhead (*Amierus nebulosus*) is of particular interest because of historical information suggesting that the local populations had been extirpated from the Woonasquatucket River (Libby, 1995). However, brown/yellow bullhead were collected in sampling areas during the fish community study, including 10 individuals in the Allendale Pond reach adjacent to the tailrace (APX1). The RIDEM survey had been restricted from boat access in the Study Area and the failure to collect bullheads may have due in part to the limited sampling that could be completed (Libby, 2001).

The following table summarizes the IBI scores between experimental sampling areas and selected background sampling areas:

Station/Background	Location	Score	Integrity Class
LPX1 / GMP1	Lower section of Lyman Mill Pond	60	Excellent
LPX1 / GMP2	Upper section of Lyman Mill Pond	46	Fair - Good
LPX1+2 / GMP1+2	Combined Lyman Mill Pond	50	Good
LPX3 / RWR1	Stream reach below Allendale dam	56	Good - Excellent
APX1 / RWR1	Stream reach near Allendale tailrace	46	Fair - Good
GMP1 / ASP1	Lower section of Greystone Mill Pond	54	Good
ASP1 / GMP1	Assapumpset Pond (Reference Area)	54	Good

The following specific study findings for the metric categories are relevant to the evaluation of effects to demersal, omnivorous fish populations:

1. Percentage white sucker – This metric distinguishes low and moderate quality water because white suckers show increased distribution of abundance despite historical degradation of surface water and they shift from incidental to dominant in disturbed sites. One location in Lyman Mill (LPX3) and one location in Allendale (APX1) received low scores for this metric.
2. Percentage omnivores – The percent of omnivores in a community increases as the physical and chemical habitat deteriorates. One location in Lyman Mill (LPX2), one location Allendale (APX1), and one location in Greystone (GMP1) received low scores for this metric.
3. Density of individuals – This metric evaluates population abundance and varies with region and stream size; generally sites with lower integrity support fewer individuals but in some nutrient poor regions, enrichment increases the density. One location in Lyman Mill (LPX2) and one location in Greystone Mill (GMP1)

received low metric scores for this metric based on comparison to background/reference areas.

In addition, the upper sampling area in Lyman Mill (LPX2), received a low score for the percent diseased individuals metric, which depicts the health and condition of individual fish. Disease, tumors, fin damage, and skeletal abnormalities are only frequently observed below point sources and in areas where toxic chemicals are concentrated (Barbour et al., 1999). A 5-year old bluegill sunfish collected at this station had a sore on the right caudal area, and a golden shiner had black spot disease. It is possible that these individuals may have been swept down from Allendale following the breaching that occurred earlier during the spring of 2001. It was also observed in the concurrent fish tissue sampling program that several large white suckers collected in Manton and all brown bullhead samples collected in Lyman Mill had numerous petechial hemorrhages (i.e., minute bloody specs) on the ventral and ventral-lateral surfaces of their bodies.

There was no indication that the demographic structure of the populations of dominant fish differed among the experimental and background/reference sampling areas (Appendix E). Mean ages for white sucker (based on scales obtained from specimens used for analytical chemistry) were 4.7, 4.7, and 5.3 years in Allendale, Lyman Mill, and Greystone, respectively. Figure 73 shows the relationship between length and weight for white sucker fish collected as part of the IBI and fish tissue sampling program – data for juvenile fish were excluded from the plot. Fish collected in both Greystone and Lyman Mill appear very similar in terms of length and weight and although the Allendale fish are considerably shorter and lighter than for either of the other two areas, the relationship between length and weight appears to hold for these samples as well. Compared to the limited demographic data obtained from the fish scales from specimens used for tissue chemistry, the majority of white suckers in Allendale is considerably smaller than in other EAs (Figure 73). At the time that the Allendale fish were collected, the Allendale Pond Dam had breached on several occasions and it is likely that the more mature (and older) white suckers had migrated either up- or down-river.

Overall, the fish assemblage within the study area appears to be in relatively good condition (Appendix E). Of the experimental sampling areas, the Allendale stream sample and the Lyman Mill upper impoundment sample appeared to be the most affected. The relative high abundance of generalist feeders, such as sunfish and bullheads, and white sucker suggest that the Allendale Reach and Lyman Mill Pond are low to moderate quality habitats, because these species tend to become dominant in disturbed ecosystems (Barbour et al., 1999).

5.3.2 Ichthyoplankton Survey

The numerically dominant ichthyoplankton larvae in Greystone Mill Pond was *Catostomus commersoni* during the first sampling round and *Lepomis* sp, followed by *C.*

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commersoni, during the second sampling round. In Lyman Mill, *C. commersoni* and *Lepomis* sp. were collected in equal abundance during the first sampling round but during the second sampling round *Etheostoma olmstedi* was the numerically dominant species, followed by *Lepomis* sp. Two species of larvae, *Lepomis* sp. and *Esox niger*, were collected in Assapumpset in May at approximately equal abundance. At the Allendale tailrace, fish larvae were also only collected in May, with *Lepomis* sp. and *Esox niger* being collected and the latter of the two in slightly higher abundance.

The majority of eggs collected across all stations during the first sampling round belonged to Centrarchidae (sunfish and bass family), with the majority being collected in Greystone. Additionally, eggs of two additional species, *Esox niger* (chain pickerel) and *Morone americana* (white perch) were also collected in Assapumpset. During the second sampling round, eggs were only collected from Greystone and were all Cyprinidae (minnow family). No eggs were collected during either sampling round in Lyman Mill or from the Allendale tailrace dipnet samples.

In an histological assessment of the ichthyoplankton samples collected during the field survey, Battelle (2003) compared the gross pathology of the field-collected ichthyoplankton with pathology observed in the ELS catfish bioassays (see Section 5.4.5) and concluded that the ichthyoplankton from the study area appeared to be developing normally with only a few gross abnormalities observed. The letter report is included in Appendix H.

The fish eggs and larvae found in Centredale Manor Study areas are typical of fish species that spawn during the season when the sampling occurred. Since sampling time was limited, only representatives of species spawning and hatching at or near the sampling time when ichthyoplankton samples were collected would be anticipated in the samples. Therefore, based on limited data, no adverse impacts of the Site on ichthyoplankton communities in Greystone, Lyman Mill, and Assapumpset were evident (Appendix E). The relative absence of fish eggs at all sampling locations other than Greystone Mill Pond could suggest that spawning was much more limited in the experimental study areas. However, the review of the developmental stages of the fish fry collected during the survey suggested that spawning was actually advanced in all sampling areas relative to Greystone (Battelle, 2003). This would explain the much lower abundance of fish eggs observed elsewhere (Battelle, 2003). Alternatively, the pathological findings that a significant percentage of the fish eggs collected from Greystone in May 2001 had various abnormalities related to necrosis of unfertilized eggs or eggs exhibiting retarded embryonic development suggest that a potential environmental stressor is present. The Smithfield WWTP, immediately upriver of the Greystone Mill Pond ichthyoplankton sampling area (GMP2), has historically been responsible for elevated nutrient and inorganic loadings to the river and is the likely source.

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5.3.3 Critical Body Residues

Tables 84 through 87 present the comparison of available NOAEL- and LOAEL-based CBRs to the demersal fish tissue CT EPCs for Assapumpset, Greystone, Allendale, and Lyman Mill, respectively. The HIs generally increase moving downriver from Greystone Mill and, for the LOAEL-based comparison, range from 9.5 (Assapumpset) to 160 (Lyman Mill). Figure 74 summarizes the CBR risk ratios by contaminant category for each exposure area. Pesticides (specifically technical chlordane) contribute most substantially to the HIs in all EAs except Assapumpset Pond and the HQs for several inorganics (including aluminum, barium, manganese, selenium, and zinc) exceed 1 across all EAs. The TCDD TEQ (dioxins/furans) exceeds the fish CBR in both Allendale and Lyman Mill; NOAEL- and LOAEL-based HQs are 2.2 and 1.5 (Allendale) and 3.7 and 2.4 (Lyman Mill), respectively. TCDD TEQs based on PCB congeners do not contribute to risk in any exposure area.

Figures 75 and 76 present the comparison of lipid-normalized TCDD TEQs (dioxin/furan and PCB congeners) to the ELS effect threshold range calculated for individual demersal fish (including brown bullhead, white sucker, and fallfish) and American eel samples, respectively. In only one instance (AP-SC-04/05) does the lipid-normalized TCDD TEQ exceed the EC/IC_{10} threshold value; this sample was a composite of two fall fish collected in April 2001. This is a conservative evaluation because lipid-normalized transfer factors between maternal fish and their eggs are typically on the order of 0.7 and the transfer factor estimated for a gravid female white sucker and her egg mass (i.e., LP-CC-07 and LP-CC-07EGG) also collected in April 2001 was 0.35 ($pg_{egg\ lipid}/pg_{fish\ lipid}$).

As discussed in Section 2.2.6, the tissue chemistry of "pre-breach" white suckers differed somewhat from those collected in July 2001 (i.e., "post-breach"); this is also evident when considering the lipid-normalized concentration data. In Allendale, the lipid-normalized TEQ (dioxin and furan congeners) was slightly higher in the pre-breach white sucker sample (0.016 ug/g_{lipid} AP-CC-06) compared to the range calculated for post-breach white sucker samples (0.006 – 0.015 ug/g_{lipid}); the TEQ (PCB congeners) was approximately an order of magnitude higher in AP-CC-06 (0.00012 ug/g_{lipid}) as compared to a range of 0.000012 – 0.000018 ug/g_{lipid} in the 3 analyzed July 2001 white sucker samples. The lipid-normalized TEQs (dioxin and furan congeners) were also slightly higher in the pre-breach white sucker samples (0.0089 and 0.013 ug/g_{lipid}, LP-CC-07 and LP-CC-08, respectively) collected from Lyman Mill compared to the range calculated for post-breach white sucker samples (0.0058 – 0.0073 ug/g_{lipid}); the TEQs based on PCB congener are approximately an order of magnitude higher in the pre-breach fish samples (0.00014 and 0.0014 ug/g_{lipid} for LP-CC-07 and LP-CC-08, respectively) as compared to a range of 0.000016 – 0.000031 ug/g_{lipid} in the 3 analyzed July 2001 white sucker samples (Figure 75).

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Lipid-normalized TCDD TEQs for American eel are all at least an order of magnitude lower than the ELS EC/IC_{10} value (Figure 76).

5.3.4 Fish Early Life Stage Test

Waterborne, laboratory exposure of channel catfish eggs to concentration mixtures of TCDD, PCB-77, and PCB-126, and the same mixtures with HCX added, negatively correlated with hatching success and survival after hatching. Lipid-normalization of the chemical data improved negative correlations and the statistical significance of concentration-response slopes, compared to non-normalized data. Details of the findings of the laboratory ELS study are presented in Appendix H.

The dose-response data were used to generate a range of Effective or Inhibition Concentrations (EC/IC) for 10, 25, 50, 75, and 90% of the exposed population. These toxic effect concentrations are presented in Tables 5-1 and 5-2 in Appendix H based on TCDD Toxic Equivalents on an egg tissue and a lipid-normalized egg concentration, respectively. The EC/IC_{10} for the TCDD, PCB-77, and PCB-126 mixture and the EC/IC_{25} for the TCDD, PCB-77, PCB-126, and HCX mixture for the fry survival at 32 day endpoint were selected as the threshold effect range. This range is 319 – 510 pg TCDD Toxic Equivalents/g egg tissue (Table 5-1, Appendix H) and 18,384 – 25,213 pg TCDD Toxic Equivalents/g egg lipid (Table 5-2, Appendix H). The EC10 value is 25% smaller (more protective) than that reported by Elonen et al., (1998) for channel catfish using similar methods. The fry survival endpoint was generally the most sensitive of those assessed during the study and has obvious population-level significance.

In the laboratory study, gross pathological legions were notable higher in fry exposed to TCDD mixtures than in the control, with data suggesting an increased incidence of developmental malformations with an increase in TCDD/PCB congener concentration (Table 3.13, Appendix H). Development abnormalities observed in the laboratory study included hemorrhages, bent tails, short tails, short snouts, and cranial malformations (Table 3.13, Appendix H). Cranial malformations, which often occur (although at a low incidence) in fish exposed to TCDD, occurred in 0 to 2% (1/51 individuals) of fish in the laboratory study and in 0 to 5% (1/21 individuals) of the fish collected from the field (Section 4, Appendix H). In general, fish collected from areas impacted by TCDD had relatively low percentages of abnormalities (0 to 6 percent) compared to TCDD-exposed fish in the laboratory study (2 to 25 percent). However, it should be noted that in the laboratory study, at higher TCDD concentrations, there was a sharp increase in the number of fry mortalities occurring within 1 week after hatching (Figure 3.6, Appendix H). In addition, as noted in the April 28, 2003 letter report by Battelle to USACE (Appendix H), the lifestages observed in the field were considerably older than those observed in the laboratory study; therefore, it is possible that abnormalities were more prevalent in the field when the fish were at an earlier stage of development, but such affected fish died prior to the sampling (Section 4, Appendix H).

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Channel catfish were used in the laboratory bioassay to develop LC_{10} and LC_{25} concentrations to protect ELS survival and reproduction in demersal fish species in the study area. The LC_{egg10} and LC_{egg50} concentrations derived for the channel catfish based on the laboratory bioassay (319 and 636 pg/g egg, respectively) are slightly less than the literature EC_{egg10} and EC_{egg50} values for this species (429 and 644 pg/g egg, respectively) (Elonen et al., 1998), indicating the fish used in the laboratory study were slightly more sensitive to TCDD. As indicated in Section 5 of Appendix H, the bioassay-derived concentrations for the channel catfish correspond to lipid-normalized concentrations of 18,384 and 26,891 pg/g lipid egg for LC_{egg10} and LC_{egg50}, respectively. In comparison, literature-derived lipid-normalized $LC_{egg}10$ values range from 1,463 pg/g lipid egg to 94,706 pg/g lipid egg, and literature-derived lipid-normalized LC_{egg50} values range from 863 to 153,529. LC_{egg10} and LC_{egg50} values for channel catfish are consequently toward the lower (more sensitive) edge of the range.

Channel catfish were selected for the bioassay due to their relative sensitivity to TCDD. Although trout have been found to be more sensitive to -TCDD, they are considered cold water fish, whereas the study area in the Woonasquatucket River is considered a warm water system. However, it should be noted that trout are stocked in the river. Based on $LC_{egg}10$ and $LC_{egg}50$ values derived for the channel catfish from the laboratory bioassay and corresponding values derived for trout species from the literature (Walker et al., 1994; Walker and Peterson, 1994; Walker et al., 1996), trout are approximately 3 to 9 times more sensitive to TCDD than channel catfish. White sucker, a common detrivorous species in the Woonasquatucket River is approximately 3 to 5 times less sensitive to TCDD than the channel catfish based on $LC_{egg}10$ and $LC_{egg}50$ values derived for the channel catfish and corresponding literature values for the white sucker (Elonen et al., 1998).

5.3.5 Risk Uncertainties

- Relatively few CBRs are available for the COPCs identified in fish tissue. Risks for those COPCs lacking CBRs were likely under-estimated. In addition, CBRs for other taxa were used when fish data were lacking and the extent to which effects can be extrapolated between different types of receptors is unknown.
- Interpretation of the ichthyoplankton study results is hampered by the failure to conduct the study during the three sampling intervals that were originally planned. The results are inconclusive because it is not possible to distinguish between absence of reproduction within an area or for a given species and improper timing for the surveys that were conducted. Although suggestive, the executed study, was unable to provide information necessary to corroborate some of the subtle malformation results obtained in the ELS study.
- The fish IBI study findings (particularly regarding tolerant/insensitive and intolerant/sensitive species) usually are the result of preferences for habitat affected by gross pollution (most usually organic carbon enrichment and low DO conditions)

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rather than the effects of toxic contaminant exposure. However, this study was supplemented with the ichthyological study that provided some insight into the effects of potential COPC exposure in the system.

- An interim TEF, or relative potency factor (RFP), of 0.0002 was established based on a biochemical competition assay that assessed the relative affinity of HCX relative to TCDD for binding to both fish (trout) and mammal (human) Ah-receptors. Only two concentrations were tested, but based on a qualitative assessment of the data, the relative affinity of HCX in these tests was estimated to be on the order of about 5000 times less than TCDD. The preliminary results, which are presented in Appendix H, did suggest that HCX competed more effectively with TCDD in the trout Ah-receptor test than with the human Ah-receptor. However, this study was not designed to evaluate the relative toxicity of HCX or to determine whether the compound is a TCDD agonist or antagonist. However, the results are supported by the limited data from *in vitro* and *in vivo* studies.
- The toxicological significance of HCX exposure to fish was not completely resolved by the ELS study. Although the conclusions that HCX is probably not as bioavailable as predicted (and may be relatively easily metabolized by fish embryos) as suggested by the study, the lack of correspondence between exposure and egg concentrations remains a question.
- The channel catfish is believed to be relatively sensitive to the effects of early life stage exposure to TCDD and dioxin-like compounds (Elonen et al., 1998). However, the relative sensitivities of many of the fish taxa in the study area have not been specifically tested and some species, including salmonids, may be considerably more sensitive than the channel catfish.
- The April 2001 breach of Allendale Pond dam resulted in a condition of greater chemical disequilibrium in the Allendale and Lyman Mill reaches. Although the pre-breach sample sizes are inadequate to draw definitive conclusions and differences in tissue lipid content, fish age, and seasonality between the before and after data sets are confounding factors, it appears that various dioxin and furan congeners became somewhat more bioavailable for some interval of time (which included the June 2001 fish sampling program) following the breach of Allendale Dam. Although co-planar PCBs were detected in most of the fish analyzed, lipid-normalized concentrations are higher in the pre-breach fish in both Allendale Pond and Lyman Mill Pond and the contribution to the overall Toxic Equivalency represented by PCBs is greater than in post-breach fish (e.g., PCBs contribute 10 percent to the Toxic Equivalency for LP-CC-08, Figure 75). In addition, pre-breach fish samples contain the highest Toxic Equivalency concentrations. Despite the likely increase in bioavailability of other dioxin and furan congeners following the Allendale Dam breach in April 2001, the lipid-normalized TCDD concentration contributes greater than 99 percent to the dioxin/furan congener Toxic Equivalency in all white sucker samples.

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5.4 RISK ASSESSMENT SUMMARY – DEMERSAL FISH

Table 88 presents the incremental risk evaluation for the demersal fish tissue/CBR comparisons conducted for Allendale and Lyman Mill based on comparison to Greystone. Incremental risks for the LOAEL-based CBRs in Allendale and Lyman Mill range from 0.035 to 110 (chlordane in Lyman Mill). Incremental risks based on LOAELs for technical chlordane, selenium, aluminum, zinc, Aroclor 1254, and TCDD TEQ (dioxins/furans) exceed one in one or both of these EAs; the incremental risk based on NOAELs range from 1.1 to 110. Of these, a clear suggestion of Site-related sourcing exists for dioxins, furans, Aroclor 1245 and possibly, chlordane (Section 2.3).

The measurement endpoints that are based on field bioassessment data explicitly consider the incremental component of risk associated with the Study area because the conclusions are made relative to the upriver background area.

The weight-of-evidence integration for the measurement endpoints evaluated for the demersal fish assessment is summarized in Table 89. The measurement endpoints related to the fish community and ichthyoplankton surveys (A_1 through A_4) do not generally suggest that the demersal fish populations are at risk within the study area. Measurement endpoint A_1, was assigned a "low-medium", while A_3, and A_4 were assigned a "medium" inference weight (Table 15); in combination they provide no compelling evidence that population or community level effects are occurring. The observation of gross lesions in brown bullhead collected from the upper portion of Lyman Mill Pond (endpoint A_2) is a weak indication of harm and this measurement endpoint is supported by additional observational data of several white suckers collected in Manton Pond as well as results obtained from the ELS study. The comparison of fish tissue concentrations to CBRs provides a strong indication of harm (endpoint B_1 assigned a "medium-high" weight). White sucker tissue concentrations exceed the CBRs for chlordane, several inorganic analytes and TCDD TEQs. However, these results are not corroborated by the site-specific tissue threshold data derived from the ELS study. Although the ELS study identified significant hatchability effects over a range of concentrations relevant to the exposures that exist within the study area, with one exception, measured fish tissue concentrations in demersal fish do not exceed the lower threshold for predicted effects in eggs even under the conservative assumption that maternal and egg concentrations are equal (on a lipid-normalized basis).

Integrating these results, it is concluded that the demersal fish community associated with the study area is not at substantial risk of harm due to the presence of contaminant release from the Centredale Manor source area. Although this analysis focused on the white sucker, a species that is known to be relatively sensitive to dioxin exposure, there remains the possibility that other components of this guild could exhibit significant dioxin-like effects at the exposure concentrations detected in the study area.

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6.0 PELAGIC FISH POPULATIONS RISK EVALUATION

This section evaluates risks to pelagic fish populations using the measurement endpoints discussed in Section 2. The measurement endpoints, sampling strategy, and methods are described, followed by an evaluation of the biological tissue data. Measured effects on pelagic fish are evaluated based on the results of fish community bioassessment and ichthyoplankton studies and comparison of measured fish tissue concentrations to CBRs. The results from the analytical chemistry and community analyses are then integrated to characterize risks to this community.

Further details of study design rationale and methods are discussed in the FSP (Harding ESE, 2001a; 2001b).

6.1 EXPOSURE ASSESSMENT - PELAGIC FISH

In this section, COPC exposure by pelagic fish is assessed using both historical and additional analytical data collected in 2001. Table A-1 summarizes the information on the analytical tissue samples for fish tissue that were used in this evaluation. These data were collected throughout the study area to represent a range of potential exposures. In addition, analytical chemistry data for biota tissue and associated sediment samples were used to develop BSAFs for predictive modeling and to develop PRGs.

All chemistry results as reported by the laboratory, including data qualifiers are presented in the DER (Harding ESE, 2003). The laboratory data were evaluated in terms of completeness, holding times, instrument performance, accuracy, precision, method reporting limits, and field quality control samples. During the quality assurance review, no data were rejected and all data were considered useable as reported.

6.1.1 Pelagic Fish Population Assessment

The fish population and community study is discussed in Section 5.1.1.

6.1.2 Ichthyoplankton Survey

The ichthyoplankton survey is discussed in Section 5.1.2.

6.1.3 Fish Tissue

Contaminant concentration data for pelagic fish tissue were collected during the field program with the same considerations used for demersal fish (see Section 5.1.3). The largemouth bass was selected for tissue collection, representing the guild of pelagic, piscivorous fish species that occur in the study area.

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Tables 90 and 91 present the fish EPC concentrations based on largemouth bass whole body tissue analytical data for the CT and RME, respectively. EPCs were developed for the Assapumpset, Greystone, Lyman Mill, and Manton EAs. No largemouth bass sample was collected in Allendale or Dyerville. Tables 82 and 83 present the fish EPCs for American eel whole body tissue analytical data for the CT and RME, respectively. EPCs for American eel were developed for all EAs except Manton.

6.2 EFFECTS ASSESSMENT - PELAGIC FISH

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

6.2.1 Pelagic Fish Population Assessment

The study results are discussed in the Section 5.2.2 and in further detail in the report provided in Appendix E.

6.2.2 Ichthyoplankton Study

The study results are discussed in the Section 5.2.3 and in further detail in the report provided in Appendix E.

6.2.3 Critical Body Residues

Section 3.2.2 describes the development of the CBRs that were used in the BERA. The NOAEL and LOAEL values for fish species presented in Table G-1 were compared to the largemouth bass whole body tissue residue data.

6.3 RISK CHARACTERIZATION - PELAGIC FISH

6.3.1 Fish IBI Study

The general finding of the fish IBI study are discussed in Section 5.3.3. The following specific study findings for the metric categories are relevant to the evaluation of effects to pelagic, piscivorous fish populations:

1. Percentage top carnivores – This metric distinguishes between systems with high and moderate water quality. One location in Allendale Pond (APX1) received a low score for this metric due to the decreased percentage of carnivores at this sample station. It is important to note that the breaching of the Allendale Dam earlier in the season probably disrupted the fish community in this exposure area and larger predatory fish may have dispersed out of the pond while the system was readjusting.

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Mean ages for largemouth bass were 6.5, 5.0, 5.2, and 4.5 years in Lyman Mill, Manton, Greystone, and Assapumpset, respectively. Figure 73 shows the relationship between length and weight for largemouth bass collected as part of the IBI and fish tissue sampling program – data for juvenile fish were excluded from the plot. Fish collected from the 4 sampling areas appear very similar in terms of length and weight, although one Lyman Mill individual is considerably lighter than would be predicted based on length. This fish was collected from the upper segment of Lyman Mill Pond and may be an individual that got swept downriver following the breach of the Allendale Pond Dam.

Overall, the fish assemblages within the study area appear to be in relatively good condition (Appendix E). Of the experimental sampling areas, the Allendale stream and the Lyman Mill upper impoundment appeared to be the most affected.

6.3.2 Ichthyoplankton Survey

The general findings of the ichthyoplankton survey are discussed in Section 5.3.4. Although no eggs or fry of the largemouth bass were collected during the survey, this is not unexpected because this species spawns relatively early in the spring (Appendix E); the larvae had probably matured to the point that they had moved into refugia and were no longer in the water column.

6.3.3 Critical Body Residues

Tables 92 through 95 present the comparison of available NOAEL- and LOAEL-based CBRs to the pelagic fish tissue CT EPCs for Assapumpset, Greystone, Lyman Mill, and Manton, respectively. The HIs are generally consistent across all EAs; for the LOAEL-based comparison, HIs range from 18 (Assapumpset) to 30 (Manton). Figure 77 summarizes the CBR risk ratios by contaminant category for each exposure area. Pesticides (specifically technical chlordane) and several inorganic COPCs (e.g., barium, mercury, selenium, and zinc) contribute most substantially to the HIs. The TCDD TEQ (dioxins/furans) is not exceeded by the LOAEL-based CBR in any exposure area and the NOAEL-based CBR is exceeded only in Lyman Mill; NOAEL- and LOAEL-based HQs in Lyman Mill are 1.3 and 0.83, respectively. TCDD TEQs based on PCB congeners do not contribute to risk in any exposure area.

Figure 78 presents the comparison of lipid-normalized TCDD TEQs (dioxin/furan and PCB congeners) to the ELS effect threshold range calculated for individual largemouth bass. The lipid-normalized TCDD TEQs exceed the EC/IC_{10} threshold value in two samples (LPX-4004 and LPX-4005). As discussed in Section 6.4.5, this is a conservative evaluation because lipid-normalized transfer factors between maternal fish and their eggs are typically on the order of 0.7 and the transfer factor estimated for a gravid female white sucker and her egg mass (i.e., LP-CC-07 and LP-CC-07EGG) also collected in April 2001 was 0.35 ($pg_{\text{egg lipid}}/pg_{\text{fish lipid}}$).

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6.3.4 Risk Uncertainties

Refer to discussion in Section 5.3.5.

6.4 RISK ASSESSMENT SUMMARY – PELAGIC FISH

Tables 96 and 97 present the incremental risk evaluation for the pelagic fish tissue/CBR comparisons conducted for Lyman Mill and Manton based on comparison to Greystone and Assapumpset, respectively. Incremental risks for the LOAEL-based CBRs in Lyman Mill and Manton based on comparison to Greystone, range from 0.077 to 10 (technical chlordane in Manton) (Table 96). LOAEL-based incremental risks for chlordane, aluminum, and barium, (the latter two in Manton only) exceed one; NOAEL-based incremental risks range from 0.32 to 20 (aluminum in Manton). The LOAEL-derived incremental risks estimates based on comparison to Assapumpset Pond are comparable, ranging from 0.092 to 7.6 (Table 97). The measurement endpoints that are based on field bioassessment data explicitly consider the incremental component of risk associated with the Study area because the conclusions are made relative to the upriver background area.

The weight-of-evidence integration for the measurement endpoints evaluated for the pelagic fish assessment is summarized in Table 98 and the results are similar to those concluded for the demersal fish population assessment. The measurement endpoints related to the fish community and ichthyoplankton surveys (A1 through A4) do not generally suggest that the pelagic fish populations are at risk within the study area. Measurement endpoints A_1 through A_4, in total provide no compelling evidence that adverse population or community level effects are occurring. The lack of observation of gross lesions in largemouth bass is also in agreement with these other measurement endpoints. The comparison of fish tissue concentrations to CBRs, which was assigned a "medium-high" inference weight, does provide a strong indication of effects (endpoint B_1). Largemouth bass tissue concentrations exceed the CBRs for chlordane, several inorganic analytes, and TCDD TEQs. However, with two exceptions, measured fish tissue concentrations in pelagic fish do not exceed the lower threshold for predicted effects in eggs even under the conservative assumption that maternal and egg concentrations are equal (on a lipid-normalized basis).

Integrating these results, it is concluded that pelagic piscivorous fish populations do present a degree of ecological integrity that is consistent with background conditions within the Woonasquatucket River, while the results of the CBR comparison (the second risk question) are inconclusive with one endpoint suggesting that fish are at substantial risk of harm and the second indicating that these risks are unlikely. In summary, the pelagic fish populations associated with the study area are not at substantial risk of harm due to the presence of contaminant release from the Centredale Manor source area.

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7.0 PISCIVOROUS MAMMAL AND BIRD POPULATIONS RISK EVALUATION

The great blue heron, kingfisher, and river otter were selected as representative species to evaluate potential risks to piscivorous species residing in the study area. Two measurement endpoints were selected to evaluate the assessment endpoint for this receptor guild including comparison of ingestion dose estimates to TRVs and TEQs and comparison of estimated piscivorous wildlife residues to CBRs. This section presents the approach used to model their exposure, the derivation of literature-based effects doses or TRVs, and the risk characterization and associated uncertainty.

7.1 EXPOSURE ASSESSMENT - PISCIVOROUS WILDLIFE

In Section 2.2.3.5, exposure profiles are presented that describe habitat use, feeding behavior, and seasonal movements of each representative species. These factors are important in regard to routes and duration of exposure. The following sections describe development of a food web model to predict dietary exposure, the estimation of piscivorous receptor tissue burdens (whole body or egg concentrations), and the data used in the assessment.

7.1.1 Food Web Modeling Approach

The objective of the food web model was to predict the exposure to COPCs that heron, kingfisher, and river otter would encounter through feeding on aquatic organisms within the study area. The food web model was used to calculate a total daily dose of the COPCs as a result of consumption by the representative species.

Tables 18 and 21 summarize the modeling inputs and exposure equations used to calculate the total daily ingested dose of each COPC. Information summarized in the Wildlife Exposure Handbook (USEPA, 1993) was used to parameterize the models; selected values are presented in Tables I-1 through I-3 (Appendix I), for the heron, kingfisher, and otter, respectively.

Due to the population focus of the assessment endpoint, TRVs were based on dietary studies that reported effects on the key demographic endpoints of mortality, growth, or reproduction. The selected ingestion toxicity data used to develop TRVs are presented in Table D-4. In addition to the ingestion rates and dietary aspects, temporal exposure and bioavailability were also considered. River otter would be expected to utilize the Site for the entire year (i.e., chronically exposed); however, the length of available shoreline associated with the reach of the Woonasquatucket River encompassing the Site is unlikely to support a breeding population of river otter. Great blue heron and kingfisher likely only use the Site during the summer and migrate during winter; however, their

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exposure duration was considered to be chronic in nature because they are on-site for at least as long as the duration of the majority of toxicity tests (particularly the reproductive studies) used to develop the TRVs for these receptors.

Dietary exposures to contaminated prey were estimated using CT EPCs for fish and invertebrates (i.e., secondary prey types) based on measured tissue concentrations in fish and invertebrates or estimated using BSAFs. The CT EPCs for brown bullhead (Assapumpset), white sucker (Greystone, Allendale, and Lyman Mill), and largemouth bass (Manton) were used to estimate dietary exposures to fish prey in these EAs. BSAFs for white sucker (Table J-4) were multiplied by the CT EPCs for sediment in Dyerville, adjusted by the percent TOC in Dyerville sediment and the Site-wide average white sucker lipid content to estimate dietary exposure from fish consumption in this exposure area.

7.1.2 Estimation of Piscivorous Wildlife Tissue Concentrations

Literature studies that measured COPC concentrations in herring gull and otter tissue and diet were used to estimate piscivorous wildlife tissue concentrations. Braune and Norstrom (1989) measured dioxin/furan and PCB congeners and various organochlorine pesticide concentrations in gull whole body, liver, and egg tissue, as well as in the tissue of alewife, one of their commonly consumed prey. Percent lipid content of each tissue type was also determined. Lipid-normalized gull egg/alewife tissue BMFs (presented in Table J-12) were calculated from information provided in Braune and Norstrom (1989). Table J-13 presents lipid-normalized BMFs for adult otter for PCB congeners and Total Aroclors presented in Leonards et al., (1997) and lipid-normalized BMFs for mink liver presented in Tillit et al. (1997) for several dioxin/furan congeners including TCDD. These BMFs were multiplied by the appropriate fish tissue CT EPCs to estimate avian piscivore egg tissue concentrations and mammalian piscivore whole body concentrations. The estimated tissue concentrations were then compared to wildlife CBRs. Tissue estimates were derived for model piscivorous avian and mammal receptors that include only fish in their diet.

Tables 99 through 104 present the estimated tissue concentrations in piscivorous bird eggs based on fish EPCs for Assapumpset, Greystone, Allendale, Lyman Mill, Manton, and Dyerville EAs, respectively. Tables 105 through 110 present similar estimates for piscivorous mammal tissue based on consuming fish in each of the EAs.

7.2 Effects Assessment - Piscivorous Wildlife

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

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7.2.1 Toxicity Reference Values

Table D-4 summarizes the selected TRVs for wildlife receptors. The TRVs are based on ingested doses (expressed as mg/kg-day) that are not anticipated to cause unacceptable survival, growth or reproductive effects. The following hierarchy, in order of decreasing preference, was used to identify toxicity values for wildlife measurement endpoint receptors: (1) chronic NOAEL; (2) subchronic NOAEL; (3) chronic LOAEL; (4) subchronic LOAEL; and (5) estimates of acute lethal does affecting 50 percent of test organisms (LD_{50}). The study exposure for laboratory toxicity tests with mammals and birds was categorized as chronic, subchronic or acute duration based on the following assumptions:

- Chronic tests involved studies that lasted longer than 180 days or, in the case of reproductive endpoints in which effects to offspring were evaluated, exposure during a critical lifestage;
- Subchronic tests lasted between 14 days and 180 days; and
- Acute tests lasted less than 14 days.

Best professional judgment was used to identify the most appropriate study and corresponding toxicity value for TRV selection when more than one toxicity study met the set of qualifying criteria applicable for study endpoint and exposure duration. The most appropriate study was the one with the least uncertainty about the accuracy of the study endpoint (USEPA, 1999). Uncertainty factors were applied, as necessary, to toxicity studies that did not report a NOAEL. The following uncertainty factors were used:

- To extrapolate between a chronic NOAEL and a chronic LOAEL, the toxicity value was multiplied by a factor of 0.1;
- To extrapolate between a subchronic NOAEL and a chronic LOAEL, the toxicity value was multiplied by a factor of 0.1; and
- To extrapolate between an acute LD50 to a chronic NOAEL, the toxicity value was multiplied by a factor of 0.02.

Wildlife TRVs based on both chronic NOAELs and LOAELs were established for all wildlife receptor COPCs.

The NOAELs and LOAELs TRVs for 2,3,7,8-TCDD were based on two studies as recommended by USEPA (1993). A study by Nosek et al., (1992) evaluated reproductive effects in ring-necked pheasant hens for a period of 10 weeks; hens were dosed weekly at

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rates equivalent to 1.4, 14, and 140 pg TCDD/g-day. A significant decrease in egg production and complete mortality in embryos was observed at the highest treatment. Based on this study, NOAEL and LOAEL TRVs for avian receptors were established at 14 and 140 pg/g-day, respectively. A three generation reproductive study of rats exposed to TCDD orally in their diet (Murray et al., 1979) was selected for development of mammal TRVs. Treatment levels in the study were 0, 1, 10, and 10 pg TCDD/g-day and significant reductions in fertility were observed in the F_1 and F_2 generations at the 10 pg TCDD/g-day level. Based on this study, NOAEL and LOAEL TRVs for mammalian receptors were established at 1 and 10 pg/g-day, respectively.

7.2.2 Critical Body Residues

Section 3.2.2 describes the development of the CBRs that were used in the BERA. The NOAEL and LOAEL values for wildlife whole body and bird eggs presented in Table G-1 were compared to the piscivorous mammal tissue and avian egg tissue residue concentrations estimated as described in Section 7.1.2.

7.3 RISK CHARACTERIZATION - PISCIVOROUS WILDLIFE

In this section the total dietary exposure is compared to NOAEL and LOAEL-based TRVs to determine the potential for adverse reproductive effects in great blue heron, kingfisher, and river otter. Using both NOAEL- and LOAEL-based TRVs provides a range of the magnitude of exceedances in order to bracket the risk uncertainties. Total dietary exposures were divided by TRVs using an HQ approach (USEPA, 1992, 1996b).

7.3.1 Food Web Modeling

Food web model exposure estimates and HQs for piscivorous wildlife are presented in Appendix K; Tables K-1 through K-6, K-7 through K-12, and K-13 through K-18 present the food web modeling results for the kingfisher, great blue heron, and river otter respectively. Figures 79 through 81 provide a summary of the LOAEL- and NOAEL-based HQs for the principal risk contributors for the great blue heron, kingfisher, and river otter, respectively. In general, the analytes presented in these figures either account for greater than 95% of the total HIs or are the only ones with HQs greater than 1. Figure 82 presents a summary of the HIs by exposure area for piscivorous wildlife receptors.

Great blue heron. Figure 79 presents a summary of the primary risk contributors for the great blue heron for each exposure area. NOAEL- and LOAEL-based HIs increase downriver of Greystone Mill Pond, with the HIs based on LOAEL TRVs ranging from 0.4 (Assapumpset) to 13.6 (Dyerville); NOAEL-based HIs range from 3.6 (Assapumpset) to 47.4 (Dyerville). Primary risk contributors include TCDD TEQs (both dioxin/furans and PCB congeners), Aroclors (particularly Aroclor-1254), DDE, and methylmercury. The elevated risk estimated for Dyerville is associated with the use of white sucker

BSAFs in lieu of measured fish tissue concentrations for this exposure area and the relatively low organic carbon content in the sediments of this reach (Table 18).

Kingfisher. Figure 80 presents a summary of the primary risk contributors for the kingfisher for each exposure area. As was the case for the great blue heron receptor, NOAEL- and LOAEL-based HIs increase downriver of Greystone, with the HIs based on LOAEL TRVs ranging from 0.7 (Assapumpset) to 25 (Dyerville); NOAEL-based HIs range from 6.17 (Assapumpset) to 85.7 (Dyerville). Primary risk contributors are the same as found for the heron.

River Otter. Figure 81 presents a summary of the primary risk contributors for the river otter for each exposure area. NOAEL- and LOAEL-based HIs appear relatively consistent throughout the Woonasquatucket River, with the HIs based on LOAEL TRVs ranging from 0.9 (Assapumpset) to 2.6 (Greystone); NOAEL-based HIs range from 3.3 (Assapumpset) to 22.6 (Dyerville). Primary risk contributors are similar to those found for the avian piscivores, with the exception that DDE does not contribute substantially to the overall HIs. The relative significance of the TCDD TEQ (dioxins/furans) increases dramatically in Allendale and Lyman Mill compared to Greystone and then decreases downriver in Manton and Dyerville.

7.3.2 Critical Body Residues

Tables 99 through 104 present comparisons of estimated COPC concentrations in avian eggs to CBRs for Assapumpset, Greystone, Allendale, Lyman Mill, Manton, and Dyerville, respectively. Tables 105 through 110 present similar comparison for a mammalian piscivore at each exposure area. Figures 83 and 84 present the results of a comparison of estimated avian piscivore egg concentrations and mammalian piscivore whole body tissue concentrations to CBRs, respectively. LOAEL-based CBR risk ratios for avian piscivore eggs range from 1.4 (Dyerville) to 34 (Manton) and NOAEL-based HIs range from 0.36 (Assapumpset) to 110 (Lyman Mill) (Figure 83). Both TCDD TEQs for dioxin/furans and PCB congeners contribute to these risk estimates. LOAEL-based CBR risk ratios for mammals range from 0.08 (Assapumpset) to 61 (Lyman Mill) based on the LOAEL CBRs and from 0.1 to 94 (Lyman Mill), based on the NOAEL CBRs.

7.3.3 Risk Uncertainties

- TCDD is a major risk contributor in the evaluation of dietary exposure and the TRVs were derived from laboratory studies with pheasant and rats (Section 7.2.1). Although these studies were well done, there is considerable uncertainty whenever laboratory results are extrapolated to different species under field conditions. There is no reason to believe that the piscivorous species that occur within the study area are more sensitive to TCDD exposures than are either the pheasant or rat, although these two species are not the most sensitive bird or mammalian species known.

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- Piscivorous wildlife tissue data were not collected to support the BERA and the derivation of estimated bird egg and mammalian whole body tissue concentrations introduces additional uncertainty into the assessment process. It is not known whether the uptake rates between alewife and gull eggs (Braune and Norstrom [1989] and between fish and otter or mink livers (Leonards et al., [1997], Tillit et al. [1997]) are appropriate for the receptors modeled in the BERA.
- Relatively few CBRs are available for the COPCs estimated in piscivorous bird egg or mammal whole body tissue. Risks for those COPCs lacking CBRs were likely under-estimated. In addition, CBRs for other taxa were used when avian/mammal data were lacking and the extent to which effects can be extrapolated between different types of receptors is unknown.
- An interim TEF, or relative potency factor (RFP), of 0.0002 was established based on a biochemical competition assay that assessed the relative affinity of HCX relative to TCDD for binding to both fish (trout) and mammal (human) Ah-receptors. Although the results are supported by the limited data from *in vitro* (mouse epithelial cell) and *in vivo* (guinea pig) studies, the actual TEF for HCX could be 1 or more orders of magnitude different than assumed in the BERA.
- Fish tissue EPCs were estimated for the Dyerville EA to evaluate dietary exposures by piscivorous wildlife rather than based on the American eel samples collected from this area. Although this was done because of concerns regarding the lack of linkage between eel tissue and the sediment concentrations where they may be collected, the use of site-derived uptake factors also introduces uncertainty into the exposure estimates.
- Fish EPCs used to model dietary exposures were based on fish samples that were relatively large (white suckers often weighed close to 2 kgs). Risk estimates for piscivorous wildlife could be biased high if the tissue concentrations of the risk contributors are lower in the size classes of fish normally consumed by these receptors. This uncertainty is likely to be more relevant for the two2 avian receptors evaluated in this assessment.
- Piscivorous wildlife exposures were estimated for each Exposure Area (delimited by existing or historical impoundments) and risk estimates derived for each EA individually. However, the SFF term used for each EA was conservatively calculated assuming that receptors that forage outside of the particular EA in question would utilize other portions of the study area (where additional contaminant exposure could occur).

7.4 RISK ASSESSMENT SUMMARY – PISCIVOROUS WILDLIFE

The incremental risk evaluation for the three modeled piscivorous wildlife receptors is presented in Tables 111 through 116.

Great blue heron. Tables 111 and 112 present the incremental risk evaluation for the great blue heron based on consumption of contaminated prey based on comparison to

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Greystone and Assapumpset, respectively. Incremental HQs (based on LOAELs) for total Aroclor exceed 1 in both Lyman Mill and Dyerville (2.4 and 11, respectively); NOAEL-based incremental risk estimates for total Aroclor range from 0.56 (Manton) to 28 (Dyerville). In addition, incremental risks (based on NOAELs) exceed 1 for total Aroclor, TCDD TEQs (dioxin/furans and PCB congeners), 4,4'-DDE, 4,4'-DDD, chromium, and zinc in at least one exposure area. The incremental risks estimates based on comparison to Assapumpset are comparable although methyl mercury is a contributor to the incremental risk estimates based on the comparison to the reference area (Table 112).

Kingfisher. Tables 113 and 114 present the incremental risk evaluation for the kingfisher based on consumption of contaminated prey based on comparison to Greystone and Assapumpset, respectively. Incremental HQs (based on LOAELs) for total Aroclor exceeds 1 in Allendale, Lyman Mill and Dyerville (1.6, 4.6 and 21, respectively); NOAEL-based incremental risk estimates for total Aroclor range from 1.1 (Manton) to 53 (Dyerville). LOAEL-based incremental risk estimates for TCDD TEQ (dioxin/furan), Aroclor 1254, and 4,4'-DDE also slightly exceed 1 in Lyman Mill, Dyerville, and Lyman Mill, respectively. In addition, incremental risks (based on NOAELs) exceed 1 for total Aroclors, TCDD TEQs (dioxin/furans and PCB congeners), 4,4'-DDE, 4,4'-DDD, chlordane, and zinc in at least one exposure area. The incremental risks estimates based on comparison to Assapumpset are comparable although methyl mercury is a contributor to the incremental risk estimates based on the comparison to the reference area (Table 114).

Otter. Tables 115 and 116 present the incremental risk evaluation for the otter based on consumption of contaminated prey based on comparison to Greystone and Assapumpset, respectively. NOAEL- and LOAEL-based incremental HQs for total Aroclor exceed 1 in Dyerville (13 and 1.3, respectively); NOAEL-based incremental risk estimates for Aroclor 1254 in Lyman Mill and Dyerville also exceed 1 (1.2 and 4.0, respectively). In addition, NOAEL-based incremental risks exceed 1 for TCDD TEQs (dioxin/furans) in all EAs except for Dyerville, and for TCDD TEQ (PCBs) in Lyman Mill only. The incremental risks estimates based on comparison to Assapumpset are comparable (Table 116).

Figure 85 presents the incremental risk estimates for piscivorous wildlife receptors for the entire Study Area.

The weight-of-evidence integration for the measurement endpoints evaluated for the aquatic invertebrate community assessment is summarized in Table 117. The measurement endpoints for dietary exposures and tissue residue (endpoints A and B, respectively) provide a strong indication of risks to piscivorous wildlife (particularly avian species).

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Integrating these results, it is concluded that piscivorous wildlife receptors may be at substantial risk of harm associated with the bioaccumulation hazard posed by dioxin, furan, and PCB congeners, DDE (avifauna only), Aroclors, and methylmercury. The observation of piscivorous avian receptors foraging in the Woonasquatucket River raises considerable uncertainty in the characterization of risks to this group due to the magnitude of the projected risks associated with embryotoxicity. It is not known whether these are self-sustaining populations or are annually supplemented by additional recruits, however.

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8.0 INSECTIVOROUS MAMMAL AND BIRD POPULATIONS EVALUATION

Tree swallows and the little brown bat were selected as representative species to evaluate potential risks to insectivorous species residing in the reach of the Woonasquatucket River and exposed to emerging insects from the aquatic habitat. Also considered in this assessment endpoint are the vermivorous American woodcock and short-tailed shrew that may be exposed to contaminants in floodplain soils associated with the river. Six measurement endpoints were selected to evaluate the assessment endpoint for this receptor guild including: (1) comparison of ingestion dose estimates to TRVs and TEQs; (2) comparison of measured (tree swallow) and (3) estimated insectivorous wildlife residues to CBRs, (4) measurement of reproductive effects in local tree swallow populations, and (5) a qualitative survey of calling amphibian. The induction of EROD activity in tree swallow nestling liver tissue was also included as a measure of exposure. Dietary exposures to adult (and nestling) swallows were estimated in the BERA because the swallow field study focused primarily on reproductive effects. This section presents the approach used to model COPC exposures by insectivorous wildlife, summarizes the tree swallow study results, and presents the risk characterization and associated uncertainties.

8.1 EXPOSURE ASSESSMENT – INSECTIVOROUS WILDLIFE

In Section 2.2.3.5, exposure profiles are presented that describe habitat use, feeding behavior, and seasonal movements of each representative species. These factors are important in regard to routes and duration of exposure. The following sections describe the estimation of insectivorous wildlife tissue (whole body or egg concentrations) and development of a food web model to predict exposure and the data used in the assessment. Finally, data used to quantify exposure, including the rationale for the data selected, are presented.

8.1.1 Food Web Modeling Approach

The objective of the food web model was to predict the exposure to COPCs that insectivorous and vermivorous wildlife receptors would encounter through feeding on insects emerging from the reach of the Woonasquatucket River or soil invertebrates that have bioaccumulated COPCs from the floodplain soils. The food web model was used to calculate a total daily dose of the COPCs as a result of consumption by the representative species. The equations used to calculate the total daily ingested dose of each COPC are provided in Tables 21 and 22 for the aquatic and floodplain habitat, respectively.

Information summarized in the Wildlife Exposure Handbook (USEPA, 1993) was used to parameterize the models; selected values are presented in Tables I-4 through I-8

(Appendix I), for the tree swallow, tree swallow nestling, little brown bat, woodcock, and short-tailed shrew, respectively. As discussed in Section 2.6, dietary exposures to nestling (as well as adult) swallows were modeled because the focus of USGS field study was on reproductive effects on the developing embryos and the USGS study did not study fledgling survival.

Due to the population focus of the assessment endpoint, TRVs were based on dietary studies that reported effects on the key demographic endpoints of mortality, growth, or reproduction. Ingestion toxicity data are presented in Table D-4.

In addition to the ingestion rates and dietary components previously discussed, temporal exposure and bioavailability were also considered. Short-tailed shrew would be expected to utilize the Site for the entire year (i.e., chronically exposed); while the Site provides breeding habitat for the other three receptors, which migrate to wintering areas every year. However, the exposure duration for all species was considered to be chronic in nature because they are on-site for at least as long as the duration of the majority of toxicity tests (particularly the reproductive studies) used to develop the TRVs for these receptors. Consistent with the USGS tree swallow study, the insectivore assessment focused primarily on the Allendale, Lyman Mill, and Manton EAs, with results compared to Greystone.

Dietary exposures to contaminated prey were estimated using CT EPCs for aquatic insects (both emerging insects and tree swallow stomach contents) and earthworms, based on measured tissue concentrations in these two prey types or estimated using BSAFs. The CT EPCs for sediment (Manton and Dyerville) were used to estimate dietary exposures to invertebrate prey in these EAs. The Site-specific BSAFs for emerging insects/tree swallow stomach contents (Table J-6) were multiplied by the sediment CT EPCs, adjusted by the percent TOC in Manton and Dyerville and the Site-wide average emerging insect lipid content to estimate dietary exposure from aquatic insect consumption in these EAs. Organic carbon and lipid content data used in the modeling are summarized in Table 19.

8.1.2 Tree Swallow Reproductive and Nestling Tissue Concentrations

EROD activity was measured in liver microsomes sampled from 12-day old tree swallow nestlings collected in 2000 and 2001. EROD activity was significantly induced in Allendale compared to 2 regional reference areas, but not so at Greystone in 2000. However in 2001, both Allendale and Lyman Mill had significantly induced EROD activity compared to both the reference areas and Greystone. EROD activity was similar between Greystone and the reference areas in eastern Minnesota and western Massachusetts in both 2000 and 2001.

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Geometric mean concentration data for TCDD demonstrates an increasing gradient of swallow embryo exposure from the upstream background (Greystone -42 pg TCDD/g) to Allendale and Lyman Mill (562 and 1,069 pg TCDD/g, respectively). The mean egg TCDD concentrations in the 4 egg samples collected from Manton Pond in 2003 is 516 pg TCDD/g.

In 2000 and 2001, Allendale and Lyman Mill had significantly higher TCDD concentrations in tree swallow nestling samples (range 570 – 990 pg/g wet weight [ww]), compared with Greystone (5.7 and 09.3 pg/g in 2000 and 2001, respectively).

At Allendale and Lyman Mill, TCDD contributed greater than 89 percent of the total TEQ (Custer and Custer, 2003); in Greystone, TCDD contributed between 42 and 62 percent of the total TEQ.

Average accumulation rate of TCDD was 147 times greater at Allendale than at Greystone in 2000 and 122 times greater in 2001 (see Figure 2 in Appendix M). In 2001, tree swallow nestlings in Lyman Mill were accumulating TCDD at a rate of 1685.9 pg/day, which was comparable with Allendale (1736.4 pg/day) and 119 times greater than at Greystone (14.18 pg/day). These uptake rates correlate well with differences in TCDD concentrations in swallow stomach content samples among study sites. In 2001, TCDD concentrations in stomach content samples from Allendale and Lyman Mill were 218.5 and 119.5 pg/g ww, respectively; the Greystone sample was only 5.66 pg/g.

8.1.3 Estimation of Insectivorous Wildlife Tissue Concentrations

Tree swallow egg and nestling tissue data were collected as part of the swallow reproductive and nestling study (Appendix M). In addition, BMFs were used to estimate the TCDD TEQs in insectivorous and vermivorous mammal tissue (i.e., little brown bat and short-tailed shrew, respectively) and in vermivorous bird eggs (i.e., woodcock). The development of BMFs is discussed in Section 7.1.2. Site-specific BSAFs were developed to estimate the uptake of TCDD TEQs based on dioxin, furan, and PCB congeners into emerging insect tissue (Tables J-6 and J-7). These BSAFs were used in conjunction with the BMFs to estimate insectivorous mammal tissue concentrations in EAs lacking measured emerging insect data (i.e., Assapumpset and Dyerville) and at Manton, where only a single swallow stomach content sample was collected in 2003.

Measured Wildlife Tissue Concentrations. Tables 118 and 119 present the CT and RME EPCs for tree swallow egg samples collected during the 4 year study in Greystone, Allendale, Lyman Mill, Manton, and a reference area, Fire Station (FST). Tables 120 and 121 present the CT and RME EPCs for tree swallow nestling samples collected in Greystone, Allendale, Lyman Mill, and Manton.

Estimated Wildlife Tissue Concentrations. Tables 122 through 127 present the estimated tissue concentrations for insectivorous mammal tissue based on estimated emerging insect concentrations for the Assapumpset, Greystone, Allendale, Lyman Mill, Manton, and Dyerville EAs, respectively. Tables 128 through 130 present the estimated tissue concentrations in vermivorous bird eggs associated with consumption of contaminated earthworms at Greystone, Allendale, and Lyman Mill, respectively; Tables 131 through 133 present similar estimates for vermivorous wildlife tissue based on earthworm consumption in each of these EAs.

8.1.4 EROD Activity in Tree Swallow Nestling Liver Tissue

EROD activity was measured in liver microsomes sampled from 12-day old tree swallow nestlings collected in 2000 and 2001. EROD activity was significantly induced in Allendale compared to 2 regional reference areas, but not so at Greystone in 2000. However in 2001, both Allendale and Lyman Mill had significantly induced EROD activity compared to both the reference areas and Greystone. EROD activity was similar between Greystone and the reference areas in eastern Minnesota and western Massachusetts in both 2000 and 2001.

The large and positive accumulation rates of TCDD in swallow nestlings (Section 8.1.2) and induction of elevated EROD activity in their livers are strongly indicative of local contamination from the individual EAs where the birds nested because of the limited foraging behaviors of the adult birds (Appendix M).

8.2 EFFECTS ASSESSMENT - INSECTIVOROUS WILDLIFE

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

8.2.1 Toxicity Reference Values

TRVs were developed as discussed in Section 7.2.1.

8.2.2 Critical Body Residues

Section 3.2.2 describes the development of the CBRs that were used in the BERA. The NOAEL and LOAEL values for wildlife whole body and bird eggs presented in Table G-1 were compared to the insectivorous and vermivorous mammal tissue and avian egg tissue residue concentrations estimated as described in Section 8.1.3.

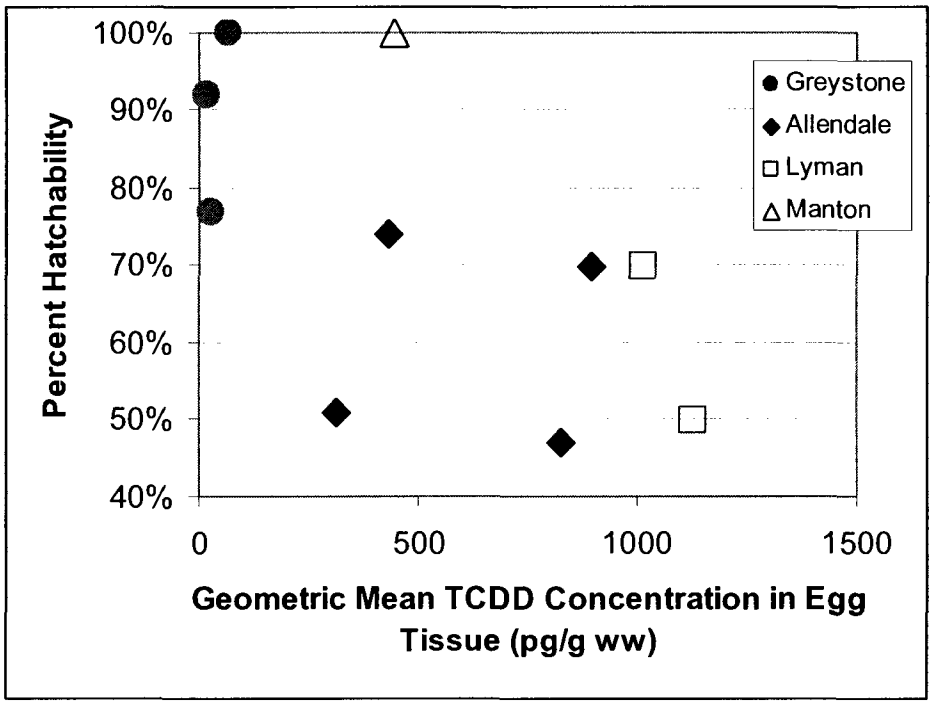
8.2.3 Tree Swallow Reproductive and Nestling Effects Study

The multi-year field study was initiated in 2000 to determine whether dioxins, furans, and other bioaccumulating compounds were being bioaccumulated and resulting in

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reproductive impairment in tree swallows that nest within the Study Area. The study design and methods are described in the study report, which is included in Appendix M.

Hatching success in Allendale and Lyman Mill were consistently poor throughout the 4 year study (Table 143). Percent hatchability at Allendale ranged between 47 and 74 percent (mean of 59%); at Lyman Mill, percent hatchability ranged from 32 to 70 percent (mean of 48%). All Manton eggs monitored in 2003 hatched successfully.



A TCDD threshold concentration of between 1,500 – 1,900 pg/g, effective at reducing tree swallow egg hatch by 50 percent effective concentration 50 (CE_{50}) was derived from this study (Custer and Custer, 2003). As a point of comparison, the chicken is believed to be the most sensitive avian receptor for TCDD toxicity with an estimated LD_{50} of between 100 and 200 pg TCDD/g (Verrett, 1976, Powell et al., 1996, Allred and Strange, 1977), which is approximately 10 times lower than obtained from this study. Laboratory-derived LD_{50}s for wild bird species such as the ring-necked pheasant (1,345 and 2,182 pg TCDD/g, for injected doses into the albumin and yolk, respectively) are comparable with the CE_{50} derived in the tree swallow study (Nosek et al., 1993).

There are relatively few studies that have assessed the effects of TCDD exposure on hatching success in natural populations of birds, and typical of field epidemiological studies, the attribution of observed effects to a single chemical stressor is usually difficult or impossible. TCDD concentrations in herring gull eggs exceeded 1,000 pg TCDD/g

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prior to 1970 when reproductive failure was commonly observed in breeding colonies in the Great Lakes; by the mid 1970s, TCDD concentrations had decreased to 160 pg TCDD/g and reproductive rates had rebounded dramatically (Gilbertson et al., 1991). White and Seginak (1994) demonstrated an inverse correlation between TCDD TEQs and nest productivity in wood duck (*Aix sponsa*) populations downstream of Bayou Meto in central Arkansas. Only 47 percent of eggs hatched from nests where the geometric mean TCDD concentration was 36 pg TCDD/g (range 1.6 – 482 pg/g ww); 79 percent of eggs hatched at the site with a geometric mean egg concentration of 4.2 pg/g (range <1 – 19) (Table G-10; Beyer et al., 1996).

An egg swapping study was also conducted in 2002 and 2003 (details described in the study reports provided in Appendix M). The study results (summarized in Table 2 of Custer, 2004), which are included in Appendix M suggest that adult swallow behavior can significantly affect hatching success rates along with the direct effects of embryological exposure to dioxins and furans. Study results from 2002 demonstrate that while only 59.7 percent of eggs collected from contaminated areas and incubated by females also nesting in contaminated areas hatched; 91.7 percent of "contaminated" eggs successfully hatched when incubated by females nesting in uncontaminated areas and only 73.3 percent of "clean" eggs successfully hatched when incubated by females that nested in contaminated areas (Custer, 2004).

8.3 RISK CHARACTERIZATION - INSECTIVOROUS WILDLIFE

In this section the total dietary exposure is compared with NOAEL and LOAEL-based TRVs to determine the potential for adverse reproductive effects in tree swallow, little brown bat, woodcock, and shrew. Using both NOAEL- and LOAEL-based TRVs provides a range of the magnitude of exceedances in order to bracket the risk uncertainties.

8.3.1 Food Web Modeling

Food web model exposure estimates and HQs for insectivorous wildlife are presented in Appendix L; Tables L-1 through L-6, L-7 through L-12, L-13 through L-15, L-16 through L-18, and L-19 through L-21 present the food web modeling results for the tree swallow, tree swallow nestling, little brown bat, woodcock, and short-tailed shrew, respectively. Figures 86 through 90 provide a summary of the LOAEL- and NOAEL-based HQs for the principal risk contributors for the great blue heron, kingfisher, and river otter, respectively. In general, the analytes presented in these figures either account for greater than 95% of the total HIs or are the only ones with HQs greater than 1. Figure 91 presents a summary of the HIs by exposure area for insectivorous wildlife receptors.

Tree Swallow. Figure 86 presents a summary of the primary risk contributors for the tree swallow for each exposure area. NOAEL- and LOAEL-based HIs are greatest at

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Allendale Mill Pond and decrease in successive downriver EAs. The HIs based on LOAEL TRVs range from 0.06 (Greystone) to 2.9 (Allendale); NOAEL-based HIs range from 0.6 to 29. For the LOAEL-based comparison, the TCDD TEQ based on PCB congener concentrations represent 60%, 2%, 14%, 0%, and 21% of the TCDD TEQ in Greystone, Allendale, Lyman Mill, Manton, and Dyerville, respectively.

Tree Swallow Nestling. Figure 87 presents a summary of the primary risk contributors for the tree swallow nestling for each exposure area. NOAEL- and LOAEL-based HIs are greatest at Allendale Mill Pond and decrease in downriver EAs. The HIs based on LOAEL TRVs range from 0.004 (Greystone) to 0.22 (Allendale); NOAEL-based HIs range from 0.04 to 2.2. For the LOAEL-based comparison, the TCDD TEQ based on PCB congener concentrations represent 54%, 2%, 15%, 0%, and 22% of the TCDD TEQ in Greystone, Allendale, Lyman Mill, Manton, and Dyerville, respectively.

Little Brown Bat. Figure 88 presents a summary of the primary risk contributors for the bat receptor for each exposure area. NOAEL- and LOAEL-based HIs are greatest at Allendale Mill Pond and decrease in downriver EAs. The HIs based on LOAEL TRVs range from 0.33 (Greystone) to 22 (Allendale); NOAEL-based HIs range from 3.3 to 220. For the LOAEL-based comparison, the TCDD TEQ based on PCB congener concentrations represent 61%, 0%, 4%, 0%, and 0% of the TCDD TEQ in Greystone, Allendale, Lyman Mill, Manton, and Dyerville, respectively.

Woodcock. Figure 89 presents a summary of the primary risk contributors for the woodcock for each exposure area. NOAEL- and LOAEL-based HIs are greatest at Greystone Mill Pond. The HIs based on LOAEL TRVs range from 3.1 (Allendale) to 6.7 (Greystone); NOAEL-based HIs range from 22 to 41. For the LOAEL-based comparison, the TCDD TEQ based on dioxin/furan congeners are less than one for all EAs and the NOAEL-based HQs are 7.1 and 2.5 for Allendale, and Lyman Mill, respectively. Several inorganic COPCs (including chromium lead, selenium, and zinc) account for the majority of the HIs and HQs are generally highest in Greystone.

Short-tailed Shrew. Figure 90 presents a summary of the primary risk contributors for the shrew for each exposure area. NOAEL- and LOAEL-based HIs are greatest at Allendale Pond. The HIs based on LOAEL TRVs range from 16 (Greystone) to 22 (Allendale); NOAEL-based HIs range from 130 to 200. With the exception of the TCDD TEQs for dioxin/furan congeners, the HQs for the primary risk contributors are greatest at Greystone.

Figure 91 summarizes the NOAEL- and LOAEL HIs for wildlife receptors by exposure area.

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8.3.2 Critical Body Residues

Tree Swallow Eggs. Tables 134 through 138 present comparisons of measured COPC concentrations in tree swallow eggs to CBRs for Fire Station (a reference area included in the 2002 study), Greystone, Allendale, Lyman Mill, and Manton, respectively. Figure 92 summarizes the results of this analysis; LOAEL-based HIs range from 0.31 (Fire Station) to 5.6 (Lyman Mill) and NOAEL-based HIs range from 0.93 to 14. TCDD TEQs (dioxin/furans) account for the majority of the risk estimates except at Greystone where pesticides are the predominant group (although no individual HQ exceeds 1). TCDD TEQs (PCB congeners) account for only 22% and 15% of the overall LOAEL-based TEQ in Allendale and Lyman Mill, respectively.

Tree Swallow Nestlings. Tables 139 through 142 present comparisons of measured COPC concentrations in tree swallow nestling tissue to CBRs for Greystone, Allendale, Lyman Mill, and Manton, respectively. Figure 93 summarizes the results of this analysis; LOAEL-based HIs range from 1.2 (Greystone) to 6.3 (Lyman Mill) and NOAEL-based HIs range from 0.44 to 13. TCDD TEQs (dioxin/furans) account for the majority of the risk estimates except at Greystone where pesticides are the predominant group (although no individual HQ exceeds 1). TCDD TEQs (PCB congeners) account for 14% and 15% of the overall LOAEL-based TEQ in Allendale and Lyman Mill, respectively.

Estimated Insectivorous Mammal Body Residues. Tables 122 through 127 present comparisons of estimated COPC concentrations in insectivorous mammals (e.g., bats) to CBRs for Assapumpset, Greystone, Allendale, Lyman Mill, Manton, and Dyerville, respectively. Figure 94 summarizes the results of this analysis; LOAEL-based HIs range from 0.20 (Greystone) to 58 (Allendale) and NOAEL-based HIs range from 0.31 to 89. TCDD TEQs (dioxin/furans) account for the majority of the risk estimates with a significant contribution from PCB congeners in Greystone and Lyman Mill. TCDD TEQs (PCB congeners) account for 52% and 13% of the overall LOAEL-based TEQ in Greystone and Lyman Mill, respectively.

Estimated Vermivorous Bird Egg Concentrations. Tables 128 through 130 present comparisons of estimated COPC concentrations in vermivorous avian (e.g., woodcock) eggs to CBRs for Greystone, Allendale, and Lyman Mill, respectively. Figure 95 summarizes the results of this analysis; LOAEL-based HIs range from 0.6 (Greystone) to 3.0 (Allendale) and NOAEL-based HIs range from 0.53 to 45.

Estimated Vermivorous Mammal Tissue Residues. Tables 131 through 133 present comparisons of estimated COPC concentrations in vermivorous mammal (e.g., short-tailed shrew) tissues to CBRs for Greystone, Allendale, and Lyman Mill, respectively. Figure 96 summarizes the results of this analysis; LOAEL-based HIs range from 1.3 (Greystone Mill Pond) to 13 (Allendale) and NOAEL-based HIs range from to 2 to 19. TCDD TEQs (PCB congeners) have a significant contribution of the TCDD TEQ in

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Greystone and Lyman Mill, accounting for 77% and 39% of the overall LOAEL-based TEQ in Greystone and Lyman Mill, respectively.

8.3.3 Tree Swallow Reproductive and Nestling Effects Study

The probability of egg hatching was significantly higher at Greystone Mill Pond than at Allendale during both 2000 and 2001. Reduced hatching was typically due to total clutch failure and hatching success was negatively correlated with TCDD egg concentrations. Forty-seven percent and 51% of eggs hatched in Allendale Pond in 2000 and 2001, whereas 92% and 77% of eggs hatched at Greystone Mill Pond (Table 143). The estimated CE_{50} for tree swallow eggs is about 1.5 to 1.9 ng/g. No significant effects on nestling growth rates or survival were noted throughout the 4-year study.

8.3.4 Qualitative Amphibian Call Survey

The amphibian call survey that was conducted during the amphibian breeding season in both 2001 and 2002 (USFWS, 2001; 2002) identified a diverse assemblage of amphibians throughout the study area. The following species were identified in one or more locations within the study area: wood frog (*Rana sylvatica*), pickerel frog (*R. palustris*), green frog, bullfrog (*R. catesbeiana*), spring peeper (*Pseudacris crucifer*), gray treefrog (*Hyla versicolor*), American toad (*Bufo americanus*), and Fowler's toad (*B. woodhousei fowleri*).

8.3.5 Risk Uncertainties

- TCDD is a major risk contributor in the evaluation of dietary exposure and the TRVs were derived from laboratory studies with pheasant and rats (Section 7.2.1). Although these studies were well done, there is considerable uncertainty whenever laboratory results are extrapolated to different species under field conditions. There is no reason to believe that the insectivorous species that occur within the study area are more sensitive to TCDD exposures than are either the pheasant or rat, although these two species are not the most sensitive bird or mammalian species known.
- Insectivorous mammal and vermivorous wildlife tissue data were not collected to support the BERA and the derivation of estimated vermivorous bird egg and mammalian whole body tissue concentrations introduces additional uncertainty into the assessment process. It is not known whether the uptake rates between alewife and gull eggs (Braune and Norstrom [1989]) and between fish and otter or mink livers (Leonards et al., [1997], Tillit et al. [1997]) are appropriate for the receptors modeled in the BERA.
- Risks for those COPCs lacking CBRs were likely under-estimated. In addition, CBRs for other taxa were used when avian/mammal data were lacking and the extent to which effects can be extrapolated between different types of receptors is unknown.

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- An interim TEF, or relative potency factor (RFP), of 0.0002 was established based on a biochemical competition assay that assessed the relative affinity of HCX relative to TCDD for binding to both fish (trout) and mammal (human) Ah-receptors. Although the results are supported by the limited data from *in vitro* (mouse epithelial cell) and *in vivo* (guinea pig) studies, the actual TEF for HCX could be 1 or more orders of magnitude different than assumed in the BERA.
- Emerging insect EPCs were estimated for the Manton and Dyerville EA using site-derived BSAFs to evaluate dietary exposures by insectivorous wildlife (e.g., bat); the use of site-derived uptake factors rather than measured tissue data introduces uncertainty into the exposure estimates.
- Insectivorous and vermivorous wildlife exposures were estimated for each Exposure Area (delimited by existing or historical impoundments) and risk estimates derived for each EA individually. However, the SFF term used for each EA was conservatively calculated assuming that receptors that forage outside of the particular EA in question would utilize other portions of the study area (where additional contaminant exposure could occur).

8.4 RISK ASSESSMENT SUMMARY – INSECTIVOROUS WILDLIFE

The incremental risk evaluation for the modeled insectivorous and vermivorous wildlife receptors is presented in Tables 144 through 151.

Tree Swallow Egg Tissue. Tables 144 and 145 present the incremental risk evaluation for tree swallow egg tissue based on comparison to Greystone Mill Pond and Fire Station, respectively. Incremental risks for TCDD TEQs (dioxins/furans) for the LOAEL-based CBRs in Allendale, Lyman Mill, and Manton, based on comparison to Greystone, are 2.3, and 3.5, and 1.6, respectively; the corresponding NOAEL-based incremental risks range from 4.7 (Manton) to 10 (Lyman Mill). Only the NOAEL-based incremental HQs for TCDD TEQs (PCB congeners) exceed 1 in both Allendale and Lyman Mill. The incremental risks estimates based on comparison to Fire Station are comparable to those for Greystone.

Tree Swallow Nestlings. Table 146 presents the incremental risk evaluation for the tree swallow nestling CBR comparison to Greystone. Incremental HQs (based on both NOAELs and LOAELs) for TCDD TEQs (dioxins/furans) exceed 1 in Allendale, Lyman Mill, and Manton, with LOAEL-based HQs ranging from 1.6 (Manton) to 3.8 (Lyman Mill); NOAEL-based HQs range from 4.4 to 11. 4,4'-DDD contributes to the incremental risk estimate in Allendale, and TCDD TEQ (PCB congeners) contributes to the incremental risks estimates for both Allendale and Lyman Mill (although only the NOAEL-based HQ for Lyman Mill (1.6) exceeds 1.

Tree Swallow. Table 147 presents the incremental risk evaluation for the tree swallow food chain exposure risk based on comparison to the Greystone background area.

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LOAEL-based incremental risk estimates for TCDD TEQs (dioxins/furans) exceed 1 in Allendale; NOAEL-based incremental risk estimates for TCDD TEQs (dioxin/furans) exceed 1 in all EAs, ranging from 1.2 (Manton) to 29 (Allendale). TCDD TEQ (PCB congeners) contributes to the incremental risk estimates for Lyman Mill although both the NOAEL- and LOAEL-based estimates are less than 1.

Tree Swallow Nestling. Table 148 presents the incremental risk evaluation for the tree swallow nestling food chain exposure based on comparison to the Greystone background area. Incremental risks were only calculated for Allendale and TCDD TEQ (dioxin/furan) is the only contributor with NOAEL- and LOAEL-based HQs of 2.1 and 0.21, respectively.

Little Brown Bat. Table 149 presents the incremental risk evaluation for the estimated bat food chain exposures based on comparison to the Greystone background area. NOAEL-based incremental risks for TCDD TEQ (dioxin/furan) all exceed 1 and range from 7.9 (Dyerville) to 220 (Allendale); LOAEL-based estimates are an order of magnitude lower. TCDD TEQ (PCB congeners) contributes to the incremental risk estimates for Lyman Mill although both the NOAEL- and LOAEL-based estimates are less than 1.

Woodcock. Table 150 presents the incremental risk evaluation for the woodcock food chain exposures based on comparison to the Greystone background area. NOAEL-based incremental risks for TCDD TEQ (dioxin/furan) exceed 1 and range from 2.0 (Lyman Mill) to 6.6 (Allendale); LOAEL-based estimates are both less than 1. Four inorganic COPCs (including cadmium, lead, selenium, and zinc) also contribute to the incremental risk estimates in Lyman Mill, although only the NOAEL-based estimates for lead and zinc exceed 1 (1.4 and 2.2, respectively).

Shrew. Table 151 presents the incremental risk evaluation for the shrew food chain exposures based on comparison to the Greystone background area. NOAEL-based incremental risks for TCDD TEQ (dioxin/furan) exceed 1 and range from 34 (Lyman Mill) to 130 (Allendale); LOAEL-based estimates are an order of magnitude lower. Three inorganic COPCs (i.e., arsenic, cadmium, and selenium) also contribute to the incremental risk estimates in Lyman Mill, with only the NOAEL-based estimates for cadmium and selenium exceeding 1 (1.5 and 4.5, respectively). The NOAEL-based incremental risk estimate for Aroclor 1254 in Allendale (1.3) also exceeds 1.

Figure 97 presents the incremental risk estimates for insectivorous wildlife receptors for the entire Study Area.

The weight-of-evidence integration for the measurement endpoints evaluated for the insectivorous wildlife assessment is summarized in Table 152. The measurement endpoints for dietary exposures, CBRs and the field tree swallow reproductive effect

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study (A, B_1, B_2, and C_1) provide a strong indication of risks to insectivorous wildlife. The results of the qualitative amphibian call surveys (endpoint C_2), although informative, were considered to be indeterminate. Although the elevated EROD activities measured in tree swallow nestling liver tissue (supported by the daily contaminant uptake data) is a strong and incontrovertible measure of exposure (endpoint D), this biomarker cannot be related directly to adverse effects as this endpoint was thus also scored as "indeterminate".

Integrating these results, it is concluded that insectivorous and vermivorous wildlife receptors are at substantial risk of harm associated with the bioaccumulation hazard posed primarily by dioxin/furan and PCB congeners. Based on the study results, insectivorous and vermivorous mammal receptors are at greatest risk; however, no population study was conducted for these receptors as was the case for the tree swallow.

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9.0 OMNIVOROUS MAMMAL AND BIRD POPULATIONS EVALUATION

The mallard and raccoon were selected as representative species to evaluate potential risks to omnivorous species residing within the study area. The single measurement endpoint selected to evaluate the assessment endpoint for this receptor guild was the comparison of ingestion dose estimates to TRVs and TEQs. No comparison of CBRs to estimated omnivorous avian egg or mammal tissue was conducted due to the uncertainties associated with tissue residues in receptors that consume a wide variety of food types. This section presents the approach used to model COPC exposures by omnivorous wildlife and presents the risk characterization and associated uncertainties.

9.1 EXPOSURE ASSESSMENT - OMNIVOROUS WILDLIFE

In Section 2.2.3.5, exposure profiles are presented that describe habitat use, feeding behavior, and seasonal movements of each representative species. These factors are important in regard to routes and duration of exposure. The following sections describe the development of a food web model to predict exposure and the data used in the assessment.

9.1.1 Food Web Modeling Approach

The objective of the food web model was to predict the exposure to COPCs that mallard and raccoon would encounter through foraging at the Site. The food web model was used to calculate a total daily dose of the COPCs as a result of consumption by the representative species.

The equations used to calculate the total daily ingested dose of each COPC are provided in Tables 21 and 22. Information summarized in the Wildlife Exposure Handbook (USEPA, 1993) was used to parameterize the models; selected values are presented in Tables I-9 through I-11 (Appendix I), for the mallard, raccoon (aquatic exposures), and raccoon (floodplain exposures), respectively.

Due to the population focus of the assessment endpoint, TRVs were based on dietary studies that reported effects on the key demographic endpoints of mortality, growth, or reproduction. Ingestion toxicity data are presented in Table D-4.

In addition to the ingestion rates and dietary components previously discussed, temporal exposure and bioavailability were also considered. The raccoon is a year round resident within the study area (i.e., chronically exposed); and although mallards may be present only during the breeding period, their exposure duration was also considered to be chronic in nature because they are on-site for at least as long as the duration of the

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majority of toxicity tests (particularly the reproductive studies) used to develop the TRVs for these receptors.

Mallards were assumed to forage on aquatic plants and invertebrates, and dietary exposures to contaminated prey were estimated using CT EPCs for crayfish or estimated using crayfish/sediment BSAFs (in Manton and Dyerville). The raccoon was assumed to forage in both the aquatic and floodplain habitats associated with the Site; prey consumed vary seasonally, with aquatic invertebrates and fish consumed primarily in the spring/early summer and fruits and soil invertebrates consumed primarily in the summer and fall. The CT EPCs for brown bullhead (Assapumpset), white sucker (Greystone, Allendale, and Lyman Mill), and largemouth bass (Manton) were used to estimate dietary exposures to fish prey in these EAs. BSAFs for white sucker (Table J-4) were multiplied by the CT EPCs for sediment in the Dyerville Reach, adjusted by the percent TOC in Dyerville sediment and the Site-wide average white sucker lipid content to estimate dietary exposure from fish consumption in this exposure area. Crayfish and earthworm CT EPCs were used to estimate dietary exposure to aquatic and floodplain invertebrate prey, respectively. Plant EPCs were estimated using literature BAFs.

9.2 EFFECTS ASSESSMENT - OMNIVOROUS WILDLIFE

The measures of effects selected to evaluate this assessment endpoint are discussed in the following sections.

9.2.1 Toxicity Reference Values

TRVs were developed as discussed in Section 7.2.1.

9.3 RISK CHARACTERIZATION - OMNIVOROUS WILDLIFE

In this section the total dietary exposure is compared with NOAEL and LOAEL-based TRVs to determine the potential for adverse reproductive effects in mallard and raccoon. Using both NOAEL- and LOAEL-based TRVs provides a range of the magnitude of exceedances in order to bracket the risk uncertainties. Total dietary exposures calculated in Section 9.1.3 were divided by TRVs using an HQ approach (USEPA, 1992, 1996b).

9.3.1 Food Web Modeling

Food web model exposure estimates and HQs for omnivorous wildlife are presented in Appendix N; Tables N-1 through N-6, N-7 through N-12, and N-13 through N-15 present the food web modeling results for the mallard, raccoon (aquatic exposures), and raccoon (floodplain exposures), respectively. Figures 98 through 100 provide a summary of the LOAEL- and NOAEL-based HQs for the principal risk contributors for the mallard, raccoon (aquatic), and raccoon (floodplain), respectively. In general, the analytes presented in these figures either account for greater than 95% of the total HIs or are the

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only ones with HQs greater than 1. Figure 101 presents a summary of the HIs by exposure area for piscivorous wildlife receptors.

Mallard. Figure 98 presents a summary of the primary risk contributors for the mallard for each exposure area. NOAEL- and LOAEL-based HIs generally decrease downriver of Greystone, with the HIs based on LOAEL TRVs ranging from 0.02 (Manton) to 0.9 (Greystone); NOAEL-based HIs range from 0.2 to 5.5. Primary risk contributors include chromium, lead, methylmercury, and zinc.

Raccoon (aquatic exposures). Figure 99 presents a summary of the primary risk contributors for the raccoon for each exposure area. NOAEL- and LOAEL-based HIs are generally comparable throughout the reach of the Woonasquatucket River evaluated, with the HIs based on LOAEL TRVs ranging from 0.6 (Assapumpset) to 1.5 (Allendale); NOAEL-based HIs range from 2.5 (Assapumpset) to 14 (Allendale). Primary risk contributors include TCDD TEQ (dioxin/furan) and Aroclors.

Raccoon (floodplain exposures). Figure 100 presents a summary of the primary risk contributors for the raccoon for each exposure area. NOAEL- and LOAEL-based HIs are greatest at Greystone and decrease in the two downriver EAs. The HIs based on LOAEL TRVs range from 0.7 (Lyman Mill) to 2.3 (Greystone) and NOAEL-based HIs range from 0.4 (Lyman Mill) to 14 (Greystone). Primary risk contributors are TCDD TEQ (dioxin/furan), Aroclor-1254, and several inorganic COPCs including arsenic, cobalt, and molybdenum.

9.3.2 Risk Uncertainties

- TCDD is a major risk contributor in the evaluation of dietary exposure and the TRVs were derived from laboratory studies with pheasant and rats (Section 7.2.1). Although these studies were well done, there is considerable uncertainty whenever laboratory results are extrapolated to different species under field conditions. There is no reason to believe that the omnivorous species that occur within the study area are more sensitive to TCDD exposures than are either the pheasant or rat, although these two species are not the most sensitive bird or mammalian species known.
- Omnivorous wildlife exposures were estimated for each Exposure Area (delimited by existing or historical impoundments) and risk estimates derived for each EA individually. However, the SFF term used for each EA was conservatively calculated assuming that receptors that forage outside of the particular EA in question would utilize other portions of the study area (where additional contaminant exposure could occur).
- Plant EPCs were estimated using uptake factors derived from the scientific literature to estimate dietary exposures by omnivorous wildlife; the use of estimated rather than measured tissue data introduces uncertainty into the exposure estimates.

9.4 RISK ASSESSMENT SUMMARY – OMNIVOROUS WILDLIFE

The incremental risk evaluation for the modeled omnivorous receptors is presented in Tables 153 through 157.

Mallard. Tables 153 and 154 present the incremental risk evaluation for mallards based on comparison to Greystone and Assapumpset, respectively. Incremental risks for the NOAEL- and LOAEL-based TRVs in Allendale, Lyman Mill, Manton Mill, and Dyerville, based on comparison to Greystone, are all less than one, as is the case with the comparison to Assapumpset.

Raccoon (aquatic exposures). Table 155 and 156 present the incremental risk evaluation for the raccoon based on comparison to Greystone and Assapumpset. The LOAEL-based incremental risk estimate for TCDD TEQ (dioxin/furan) slightly exceeds 1 in Allendale; with NOAEL-based incremental risks ranging from 1.2 (Dyerville) to 12 (Allendale). Total Aroclors and Aroclor 1254 also contribute to the incremental risk estimates for Dyerville but only the NOAEL-based estimate for total Aroclor exceeds 1. The incremental risks to raccoon are similar in the comparison with Assapumpset (Table 156).

Raccoon (floodplain exposures). Table 157 presents the incremental risk evaluation for the raccoon based on comparison to Greystone. Incremental risks for the NOAEL- and LOAEL-based TRVs in Allendale and Lyman Mill, based on comparison to Greystone, are all less than 1.

Figure 102 summarizes the incremental risk estimates for omnivorous wildlife receptors.

The weight-of-evidence integration for the measurement endpoints evaluated for the omnivorous wildlife assessment is summarized in Table 158. Omnivorous mammal and bird populations that forage within the study area are not at substantial risk of harm due to exposure to Site-related contaminants in surface water, floodplain soil, and terrestrial prey items. However, omnivorous mammals could be adversely affected as a result of exposure to Site-related contaminants in sediment and aquatic prey. Although the exclusive use of aquatic habitat by omnivorous mammals, such as the raccoon, could result in substantial population-level effects, the spatially and temporally varied diets and exposures of these receptors minimizes the likelihood that demographically significant effects would occur.

Integrating these results, it is concluded that omnivorous wildlife receptors are not at substantial risk of harm associated with the dietary exposures to COPCs in the study area.

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10.0 ECOLOGICAL RISK UNCERTAINTY ANALYSIS

This section includes a discussion of major limitations of the analyses, any sources of uncertainties, and, if possible, any indication as to whether these uncertainties and limitations may have resulted in an over- or under-estimation of risk. The uncertainty section may also include unusual site conditions or extenuating circumstances that may be pertinent to risk management decisions. Uncertainties in the quantification of risk associated with the site are identified and their impacts on risk estimates are discussed below.

Uncertainties associated with the selection of COPCs, exposure assessment, effects assessment, and overall risk characterizations are discussed. Table 159 summarizes the principal risk uncertainties and identifies the relevant assessment endpoint(s) and COPCs, as well as the direction (i.e., under- or over-estimating risks) and magnitude of likely effect. Those uncertainties that likely resulted in the BERA risk estimates being substantially under-estimated are indicated by shading: these include: (i) not deriving TCDD plant tissue concentrations to evaluate omnivorous mammal risks, (ii) not characterizing floodplain exposures in the Manton and Dyerville EAs, (iii) the lack of critical body residue data for certain COPCs, and (iv) the limited ichthyological survey results. On the other hand, the following categories probably resulted in risk estimates generally being over-estimated in the BERA: (i) estimated fish tissue EPCs for Dyerville, (ii) the use of relatively large fish in the tissue sampling program, and (iii) the use of standard benchmarks as one line of evidence for several of the assessment endpoints.

10.1 SELECTION OF COPCS

The extensive environmental investigations have provided a substantial body of information that has been utilized in the BERA. The identification of COPCs was done in a conservative manner that assured that the quantification of risk to ecological receptors included all relevant potential stressors. Based on currently available information, it is unlikely that any detected substances that were not selected as COPCs would have a substantial impact on the BERA results and conclusions if they had been retained in the analysis.

Background conditions were not specifically considered in the selection or elimination of substances as COPCs. Several of the persistent organic COPCs (such as dioxins, furans, and PCBs), while they are not naturally-occurring substances, are detectable at some concentration almost ubiquitously in environmental samples such as biota and sediments. Therefore, exposure concentrations of those COPCs represent "total" exposure potential from both site-related and non-site-related sources.

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10.2 EXPOSURE AND MODELING UNCERTAINTIES

Uncertainties associated with the exposure assessment step of the BERA affect the interpretation of the significance of the HQs. The main uncertainties are associated with the development of EPCs, receptor selection, exposure parameters, estimation of bioaccumulation factors for various environmental media, and the evaluated exposure pathways.

10.2.1 Development of Exposure Point Concentrations

As shown in Table 1 of the Data Evaluation Report (MACTEC, 2003), collection of 10 fillet and 10 offal samples of largemouth bass in each of the following areas had been planned: Allendale, Lyman Mill, Greystone, and Assapumpset. Three fillet and three offal samples of largemouth bass had been proposed for Manton and Dyerville. However, because a further breach of the Allendale Dam occurred and the water level in Allendale Pond receded just prior to the 2001 fish tissue sampling program, no largemouth bass samples could be collected from this EA. At Assapumpset, four rather than 10 largemouth bass samples were collected despite extensive sampling effort. No largemouth bass sample was obtained from Dyerville. Three American eel whole body samples were collected from Dyerville Pond as a contingency.

There is some interpretive uncertainty associated with several fish tissue samples that were included in the data set used to derive fish tissue EPCs for Allendale and Lyman Mill. The samples were collected in April 2001 prior to the summer 2001 biota data collection program. Although the white sucker samples are limited in number and no other fallfish were collected as a point of comparison, these samples do provide a snapshot of conditions at a time just prior to complete breaching of the Allendale Dam in the early summer of 2001. The samples included white sucker and fallfish collected from Allendale and Lyman Mill. The following information concerning these April 2001 fish samples is taken from Task 22F ELS Fish/Egg Tissue Chemistry Data Report, Post-Third Party Validation (Battelle, Feb. 7, 2002).

- Although five fallfish were collected from Allendale Pond, per USEPA direction, samples AP-SC-01 and AP-SC-02 were composited, and that composite sample was composited with AP-SC-03 to form the composite sample YU40COMP + W2030. Samples AP-SC-04 and AP-SC-05 were composited per USEPA direction to form composite sample YU41COMP. One whole body White Sucker sample was collected from Allendale Pond (AP-CC-06).

- One female white sucker was collected from Lyman Mill. The eggs were removed for analysis and the remaining whole body (minus the eggs) was submitted for analysis (LP-CC-07, LP-CC-07 EGG). One additional whole body male white sucker sample was collected from Lyman Mill (LP-CC-08).

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The April 2001 fish samples were analyzed for a limited analytical suite that included only dioxins and furans and PCB congeners.

The single April 2001 white sucker whole body sample from Allendale Pond is much smaller (only 90 grams or roughly 3 ounces) and most likely younger than the other ten white sucker samples collected in summer 2001 from Allendale Pond (weights ranged from 900 grams to 1450 grams or roughly 2 pounds to 3.2 pounds). Thus, the April 2001 Allendale white sucker is smaller (only one tenth the size of all other samples) and younger than any of the existing samples from Allendale.

One white sucker sample from Lyman Mill Pond from April 2001 (LP-CC-08) has a substantially higher concentration of PCB-congener dioxin equivalents (0.976 ug/kg) than any of the other white sucker samples collected from Lyman Mill (mean of 3 post-breach samples is 0.0401 ug/kg and the maximum is 0.0503 ug/kg). The other April 2001 Lyman Mill Pond White Sucker sample (LP-CC-07) has 0.137 ug/kg of the PCB-congener dioxin equivalents. Neither of the April 2001 sample concentrations are more than two orders of magnitude higher than the post-breach concentrations, but both are higher than the maximum concentration.

The dioxins/furans TEQ for these two April 2001 white sucker samples (0.423 ug/kg and 0.457 ug/kg are reasonably consistent (perhaps somewhat lower) with the post-2001 breach samples from Lyman Mill (mean of 0.580 ug/kg and maximum of 1.37 ug/kg). The April 2001 Lyman Mill white sucker samples appear to have higher PCB TEQs (number of samples is very small) and lower dioxins/furans TEQs than do the post-2001-breach white sucker samples.

Another consequence of the breaching of the Allendale Pond Dam is the likelihood that a substantial mass of dioxin-contaminated sediments was transported from Allendale to the upper portion of the Lyman Mill exposure area. The sediment EPCs developed in the BERA relied primarily on analytical data collected prior to the dam breach and as a result, ecological exposures in Allendale may have been over-estimated and those in Lyman Mill under-estimated in the assessment.

These data support the suggestion that the April 2001 breach of Allendale Pond dam resulted in a condition of greater chemical disequilibrium in the Allendale and Lyman Mill reaches. Although the pre-breach sample sizes are inadequate to draw definitive conclusions and differences in tissue lipid content, fish age, and seasonality between the before and after data sets are confounding factors, it appears that various dioxin and furan congeners became somewhat more bioavailable for some interval of time (which included the June 2001 fish sampling program) following the breach of Allendale Dam. Although co-planar PCBs were detected in most of the fish analyzed, lipid-normalized concentrations are higher in the pre-breach fish in both Allendale Pond and Lyman Mill

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Pond and the contribution to the overall Toxic Equivalency represented by PCBs is greater than in post-breach fish (e.g., PCBs contribute 10 percent to the Toxic Equivalency for LP-CC-08, Figure 75). In addition, pre-breach fish samples contain the highest Toxic Equivalency concentrations.

Despite the likely increase in bioavailability of other dioxin and furan congeners following the Allendale Dam breach in April 2001, the lipid-normalized TCDD concentration contributes greater than 99 percent to the dioxin/furan congener Toxic Equivalency in all white sucker samples.

The representative nature of the fish tissue data for the four EAs may not be equivalent. For Allendale, data for 20 whole body fish tissue samples were used in the BERA. For Lyman Mill, data for twenty whole body fish tissue samples were utilized. However, for Manton, data for only three whole body fish tissue samples, and in Dyerville, BSAFs were used to estimate EPCs for white sucker. Therefore, based on the number of samples available and the number of species represented, it appears that the results of the BERA for piscivorous wildlife exposure in Allendale and Lyman Mill are based on more representative data (more certain) than are the results for the evaluation for Manton and Dyerville. The BERA also used site-specific BSAFs to estimate fish (i.e., white sucker) tissue EPCs in Dyerville rather than measured concentrations in American eel. This was done to improve the consistency of the evaluation with other EAs where white sucker tissue data were available and also in consideration of the mobile nature of adult American eel that can readily move between impoundments. However, the measured American eel tissue concentrations of dioxins and furans (and the derived TEQs) were typically an order of magnitude lower than in white sucker for samples collected in the same exposure area. Consequently, risks to piscivorous and omnivorous wildlife exposed in Dyerville may have been over-estimated in the BERA. The relatively low organic carbon in Dyerville sediments contributed to the elevated piscivorous risk estimates because of the use of the white sucker BSAFs for this EA.

Data were not available for all potential dietary components of these species, and with two exceptions, exposures to fish tissue were estimated using measured concentrations in white sucker. In general, tissue concentrations in white sucker were higher than largemouth bass, however, largemouth bass tissue data were used to estimate dietary exposures in Manton because no white suckers were collected during the 2001 field sampling event. BSAFs were used to estimate fish EPCs for piscivores dietary exposure and the estimated fish tissue concentrations are greater than average COPC concentrations elsewhere because the average sediment TOC at Dyerville is the lowest of any EA.

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10.2.2 Selection of Receptors of Concern

Receptor species were selected that would be expected to receive elevated exposures to contaminants that bioaccumulate in aquatic and floodplain habitat. Although considerable time was spent by trained biologists in the study area during the conduct of the 2001 field sampling program, no quantitative census of wildlife populations was conducted. Several of the selected receptors were routinely observed at the Site (including kingfisher, great blue heron, tree swallow, and raccoon) but the presence of large piscivorous mammals was not verified. It is possible that the available habitat, which is located within an urban suburb, is not capable of supporting a wide-ranging species such as the river otter.

The BERA did not specifically evaluate plant species as a receptor group; however, there is no indication that plants are sensitive to TCDD and other COPCs with dioxin-like effects.

10.2.3 Exposure Parameters

The relationship between receptor size and dietary intake is a critical factor in estimating exposure. In addition, dietary composition affects exposure because different food sources contain varying levels of COPCs. Although literature data exist for dose calculation inputs such as body weight, ingestion rate, and dietary composition for the measurement endpoint receptors evaluated in the BERA, natural populations may exhibit considerable variability in these parameters. Uncertainty is inherent in the use of these values in part because they were derived from literature sources rather than being empirically measured at the Site. Therefore, use of literature-derived exposure parameters increases uncertainty that could result in an over- or underestimation of the typical actual exposures encountered by receptors at the Site. The wildlife exposure models were parameterized using available information for adult females for each selected receptor species and average values were selected for the parameter values where a range of data were provided.

Wildlife exposures were estimated for each Exposure Area (delimited by existing or historical impoundments) and risk estimates derived for each EA individually. However, the SFF term used for each EA was conservatively calculated assuming that receptors that forage outside of the particular EA in question would utilize other portions of the study area (where additional contaminant exposure could occur). This assumption could have resulted in risks being over-estimated somewhat.

10.2.4 Bioaccumulation Factors

Site specific prey tissue data were available to estimate exposures to most of the wildlife receptor species evaluated in the BERA and use of this information minimized

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uncertainties associated with this aspect of exposure estimation. Derived BSAFs were used to estimate fish tissue concentrations in Dyerville and emerging insect tissue concentrations in Manton, Dyerville, and Assapumpset EAs.

Literature uptake factors were used to estimate plant tissue concentrations because no site-specific tissue data were collected. There is greater attendant uncertainty with the use of literature values. In addition, although 2,3,7,8-TCDD is known to bioaccumulate in the tissue of some aquatic plant species (e.g., Yockim et al., 1978), no attempt to model the uptake of dioxin, furan, or PCB congeners into plant tissue was made in the wildlife exposure modeling. As a result, dietary exposures to omnivorous wildlife (e.g., mallard and raccoon) were underestimated.

Another consequence of the Allendale Pond Dam breach that occurred shortly before the collection of biological tissue samples in June 2001, is that the river system was temporarily farther from "stable" conditions than had existed previously. The resuspension and downriver migration of contaminated-bound sediment likely resulted in a substantial increase in the bioavailability of COPCs. The consequences of this event on the tissue residues of fish collected in Lyman Mill later that year are not known.

10.2.5 Evaluated Exposure Pathways

No evaluation of potential ecological exposure to surface water or floodplain soil in Manton or Dyerville was conducted because of a lack of analytical data.

10.3 ECOLOGICAL EFFECTS

10.3.1 Toxicity Reference Values

Potential uncertainties are related to the appropriateness of literature-derived toxicity data. TRVs used in the BERA are based on an extensive search of both primary peer-reviewed literature and secondary literature, such as government reports and technical conference proceedings. The number and types of information sources reviewed is believed to be adequate to capture the majority of relevant sources of ecotoxicological literature. Very limited data exist for specific wildlife receptors or guilds (e.g., omnivorous birds, herbivorous mammals, soil invertebrates) in the toxicological literature. Therefore, TRVs were selected from toxicity data and applied to broad classes of animals, and used to quantify risk for specific measurement endpoint receptors.

Chronic toxicological data were selected preferentially in developing TRVs. However, available toxicological data are not always associated with chronic exposure duration. Therefore, there are uncertainties in extrapolating the results of shorter term exposures to the chronic exposures assumed for receptors at the Site. Chronic NOAELs were the preferred toxicity endpoint for selection of TRVs; however, ecological toxicity data were

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limited form some COPCs and some wildlife guilds. Therefore, other endpoints (for example, subchronic NOAELs, LC_{50} values were selected for use as TRVs. When an endpoint other than a chronic NOAEL was selected as a TRV, an uncertainty factor was applied to the reported value to provide an additional level of conservatism in the risk estimation process. Application of conservative application factors may result in risks being overestimated.

There is little or no toxicological data available for some COPCs. Therefore, when appropriate, toxicity data for similar chemicals were used as surrogates for the COPCs. For example, oral ingestion toxicity data for benzo(a)pyrene were used to derive TRVs for many of the PAH compounds lacking relevant information. Use of such surrogate toxicity data may underestimate or overestimate risks from the COPCs. In some cases, no appropriate toxicity or surrogate data were identified in the toxicological literature. For instance, there was no information available to establish CBRs for some of the analytes detected in Site fish tissue. As a result, risk from some COPCs could not be quantified. This may underestimate ecological risks at the Site.

In general, uncertainty is also associated with the extrapolation of literature-derived toxicity endpoints (especially laboratory-based studies) to equivalent endpoints for measurement endpoint receptors at the Site because of discrepancies in exposure conditions. The majority of the toxicity data evaluated and used in the BERA were derived from laboratory studies. Laboratory settings do not necessarily mimic field conditions and exposures and typically are designed to control various factors in order to isolate one parameter in particular. Although controlled experiments result in a more valid interpretation of the isolated parameters, uncertainty is associated with the assumption that field exposures are equivalent to laboratory exposure conditions.

10.3.2 Dioxin, Furan, and PCB Congeners

Wildlife exposures to dioxin, furan, and PCB congeners were estimated using TRVs similar to those recommended in USEPA (1993) and the consensus-based TEFs from Van den Berg et al. 1998. This approach represents the most recent risk assessment approach for evaluating dioxins and furans. This approach has been employed because there is not adequate toxicity testing for each of the hundreds of dioxin and furan congeners. Although the TEFs do have scientific basis, the use of the TEFs to estimate the ecological effects of each of the congeners does have some uncertainty associated with it. However, the predominant congener in environmental media at the four exposure points is 2,3,7,8-TCDD. Since 2,3,7,8-TCDD is considered to be the most toxic of the dioxin and furan congeners, the use of the TEFs has less impact on the risk assessment than would be the case where 2,3,7,8-TCDD was not the predominant congener.

The use of TEFs does not account for antagonistic or synergistic interactions between congeners and therefore may underestimate the risk to wildlife. The TEQ approach also

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does not account for toxicity of dioxin/furan and PCB congeners that have a non AhR-mediated toxicological mechanism. However, risk associated with exposure to compounds that exhibit non-dioxin like effects was separately considered using toxicological data for PCB mixtures.

Although the SERE (TetraTech, 1997) assumed that HCX was as potent as TCDD based on structural similarities between these two compounds, several empirical studies suggest that HCX poses a much less serious environmental hazard. As described in Section 2.4.1.3, the BERA used an interim TEF of 0.0002 for HCX based on results from Hahn (2001), who conducted an *in-vitro* AhR binding affinity study using both fish and mammalian receptors. The relative low toxicity of HCX (compared to TCDD) is further supported by an *in-vitro* study of mouse epithelial cell growth inhibition (Viswanathan et al., 1987) and a 42-day *in vivo* study using guinea pigs (DeCaprio et al., 1987), which suggest that HCX is between 10^{-6} and $2x10^{-5}$ less toxic than TCDD, respectively. These two studies, conducted at the cytological and organism levels of biological organization, provide information that corroborates the biochemical assay results conducted by Hahn (2001).

10.4 RISK CHARACTERIZATION UNCERTAINTIES

In addition to the uncertainties introduced as a result of the data availability and representativeness issues discussed above, the main sources of uncertainty associated with risk characterization involve the interpretation of HQ results. Due to the conservative assumptions that were made in developing both exposure and effects assessments, HQs that are greater than 1 do not indicate that a substantial population- or community-level harm has occurred. Attempts were made to bound these uncertainties in the BERA by considering risk associated with both chronic NOAELs and LOAELs and evaluating risks under both RME and CT exposure assumptions.

The inclusion of several field studies that were focused on community-level (e.g., fish IBI, ichthyoplankton, macroinvertebrate assessments) or population level (e.g., swallow reproductive study) endpoints minimized the typical uncertainties associated with translating between measures of effect and the assessment endpoints.

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11.0 RISK ASSESSMENT CONCLUSIONS

The potential risks associated with exposures for ecological receptors that utilize the aquatic and adjacent floodplain habitat of the Woonasquatucket River were characterized. The risk characterization included the evaluation of six assessment endpoints and included the evaluation of direct contact with surface water and sediment and the consumption of contaminated biota in four EAs (Allendale, Lyman Mill, Manton, and Dyerville). The assessment endpoint receptors evaluated included:

- Aquatic and floodplain invertebrate communities
- Demersal, omnivorous fish populations
- Pelagic, piscivorous fish populations
- Piscivorous wildlife populations
- Insectivorous wildlife populations
- Omnivorous wildlife populations.

Risks were primarily calculated using CT EPCs although RME exposures were considered in the evaluation of certain endpoint receptors. Both NOAEL- and LOAEL-based TRVs were employed to quantify uncertainty associated with the risk estimates. The calculated risks were compared to a Hazard Index (HI) value of 1. Table 160 summarizes the results of the BERA for each of the assessment endpoints.

The following conclusions were drawn from the integration of lines of evidence in the ecological risk analysis.

- The benthic macroinvertebrate community that occurs in low-gradient habitats (lentic) upstream of impoundments within the study area is at substantial risk of harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.
- However, the discharge of VOC-contaminated groundwater into the reach of the Woonasquatucket River adjacent to the Site does not appear to have adversely affected the benthic macroinvertebrate community associated with lotic (riffle/run) habitats.
- The soil invertebrate community occurring within the Woonasquatucket River floodplain at the study area is not likely at substantial risk of harm due to exposure to Site-related contaminants in floodplain soil or tissue residues.
- Demersal, omnivorous fish populations that occur within the study area may be at substantial risk of harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.

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- Pelagic, piscivorous or semi-piscivorous fish populations that occur within the study area may be at substantial risk of harm due to exposure to Site-related contaminants in surface water, sediment, and tissue residues.
- Piscivorous mammal and bird populations that occur within the study area may be at substantial risk of harm due to direct exposure to Site-related contaminants in surface water, sediment, and prey. In addition, consumption of contaminated fish may result in elevated tissue residues in these receptors resulting in adverse reproductive effects (i.e., bioaccumulation hazard).
- Dioxins and furans (particularly 2,3,7,8-TCDD), coplanar PCBs, Aroclor-1254, 4,4'-DDE, and methylmercury are the largest contributors to the estimated risk for piscivorous wildlife species.
- Insectivorous and vermivorous mammal and bird populations that occur within the study area appear to be at substantial risk of harm due to direct exposure to Site-related contaminants in surface water, sediment, floodplain soil and prey. In addition, consumption of contaminated insect and earthworm prey may result in elevated tissue residues in these receptors resulting in adverse reproductive effects (i.e., bioaccumulation hazard).
- Dioxins and furans (particularly 2,3,7,8-TCDD) and coplanar PCBs are the primary contributors to risks for insectivorous wildlife that consume insects emerging from the Woonasquatucket River.
- Dietary exposures and measured (or modeled) tissue burdens in avian eggs and mammal whole body tissue pose a substantial risk of harm to piscivorous and insectivorous wildlife species.
- Omnivorous mammal and bird populations that forage within the study area are not at substantial risk of harm due to exposure to Site-related contaminants in surface water, floodplain soil, and terrestrial prey items. However, omnivorous mammals could be adversely affected as a result of exposure to Site-related contaminants in sediment and aquatic prey. Although the exclusive use of aquatic habitat by omnivorous mammals, such as the raccoon, could result in substantial population-level effects, the spatially and temporally varied diets and exposures of these receptors minimizes the likelihood that demographically significant effects would occur.

The concentrations in biota tissue of the predominant risk contributors are directly related to corresponding sediment concentrations. Although direct contact exposures to sediments are not associated with the largest risks at the Site, the sediments appear to be associated with the largest risks at the Site.

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ACRONYMS

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LIST OF ACRONYMS AND ABBREVIATIONS

AQUIRE	Aquatic Information Retrieval
AhR	Ah receptor
APB	Allendale Pond exposure area
AVS	acid volatile sulfide
AWQC	Ambient Water Quality Criteria
BERA	Baseline Ecological Risk Assessment
BHHRA	Baseline Human Health Risk Assessment
BSAF	biota-sediment accumulation factor
BMFs	biomagnification factors
CE_{50}	effective concentration 50
COIs	chemicals of interest
COPCs	chemicals of potential concern
CBRs	critical body residues
CSM	conceptual site model
CSO	combined sewer outfall
CT	central tendency
DDD	dichlorodiphenyldichloroethane
DDE	dichlorodiphenyldichloroethylene
DDT	dichlorodiphenyltrichloroethane
DER	Data Evaluation Report
DQOs	data quality objectives
DYP	Dyerville Reach exposure area
EAs	Exposure Areas
EC/IC_{10}	effect/inhibition concentration 10
EC/IC_{25}	effect/inhibition concentration 25
ECOSAR	Ecological Structure Activity Relationship
ECOTOX	ECOTOXicological database
EE/CA	Engineering Evaluation/Cost Analysis
Eh	redox potential
ELS	early life stage
EPC	exposure point concentration
EPT	Ephemeroptera, Plecoptera, and Trichoptera
ER-L	Effects Range-Low
EROD	ethoxyresorufin-O-deethylase
ESGs	equilibrium partitioning-based sediment guidelines
ET	ecotoxicity threshold

FSP	Field Sampling Plan
FST	Fire Station reference area
GLEC	Great Lakes Environmental Center
GMP	Greystone Mill Pond background area
HCX	hexachloroxanthene
HI	hazard index
HQ	hazard quotient
IBI	Index of Biotic Integrity
INCHEM	International Chemical Database
IPCS	International Programme on Chemical Safety
IRIS	Integrated Risk Information System
LD_{50}	lethal dose 50
LELs	lowest effect level
LOAEL	lowest observable adverse effect level
LOECs	lowest observed effect concentration
LPX	Lyman Mill Pond exposure area
MAP	Manton Pond exposure area
NOAA	National Oceanic and Atmospheric Administration
NOAEL	no observed adverse effect level
NTCRA	Non-Time-Critical Removal Action
NWQC	National Water Quality Criterion
OME	Ontario Ministry of the Environment
ORNL	Oak Ridge National Laboratories
PAHs	polycyclic aromatic hydrocarbons
PCBs	polychlorinated biphenyls
PCDDs	polychlorinated dibenzodioxins
PCDFs	polychlorinated dibenzofurans
ppt	parts per trillion
PRGs	preliminary remediation goals
RAB	Assapumpset Pond/Brook reference area
RIDEM	Rhode Island Department of Environmental Management
RIPDES	Rhode Island Pollutant Discharge Elimination System
RME	reasonable maximum exposure

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SEM	Simultaneously Extracted Metals
SERE	Streamlined Ecological Risk Evaluation
SQC	sediment quality criteria
SOW	Statement of Work
SVOCs	semivolatile organic compounds
TCDD	tetrachlorodibenzo-p-dioxin
TCX	tetrachloroxanthene
TEC	toxic equivalent concentration
TEF	toxicity equivalent factor
TEQ	toxic equivalent quotient
TMDL	total maximum daily load
TOC	total organic carbon
TRV	toxicity reference value
UCL	upper concentration limit
USACE	U.S. Army Corps of Engineers, New England District
USEPA	U.S. Environmental Protection Agency
USFWS	U.S. Fish and Wildlife Service
VOCs	volatile organic compounds
WWTP	wastewater treatment plant

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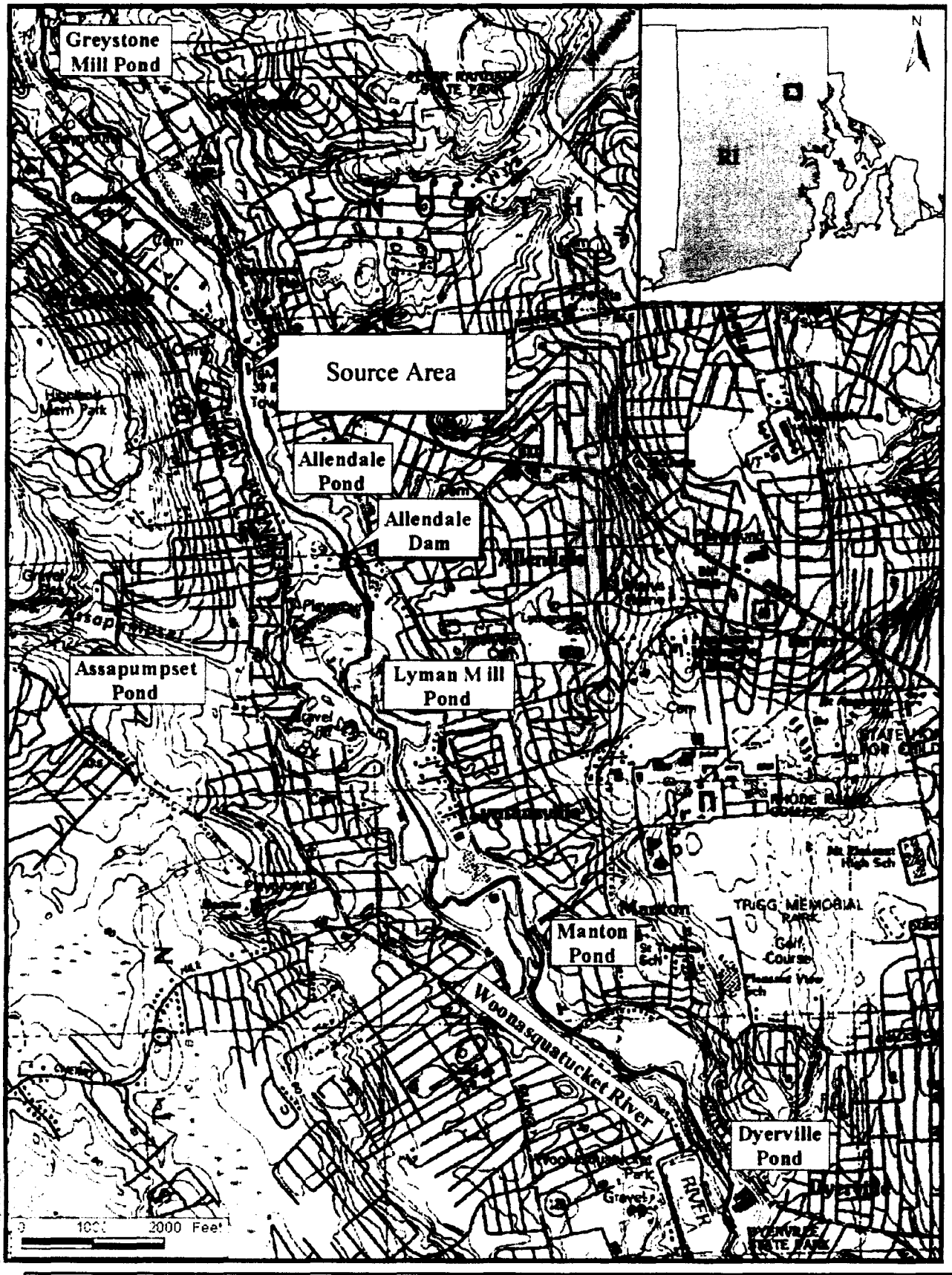
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Base map source Battelle Ocean Sciences.

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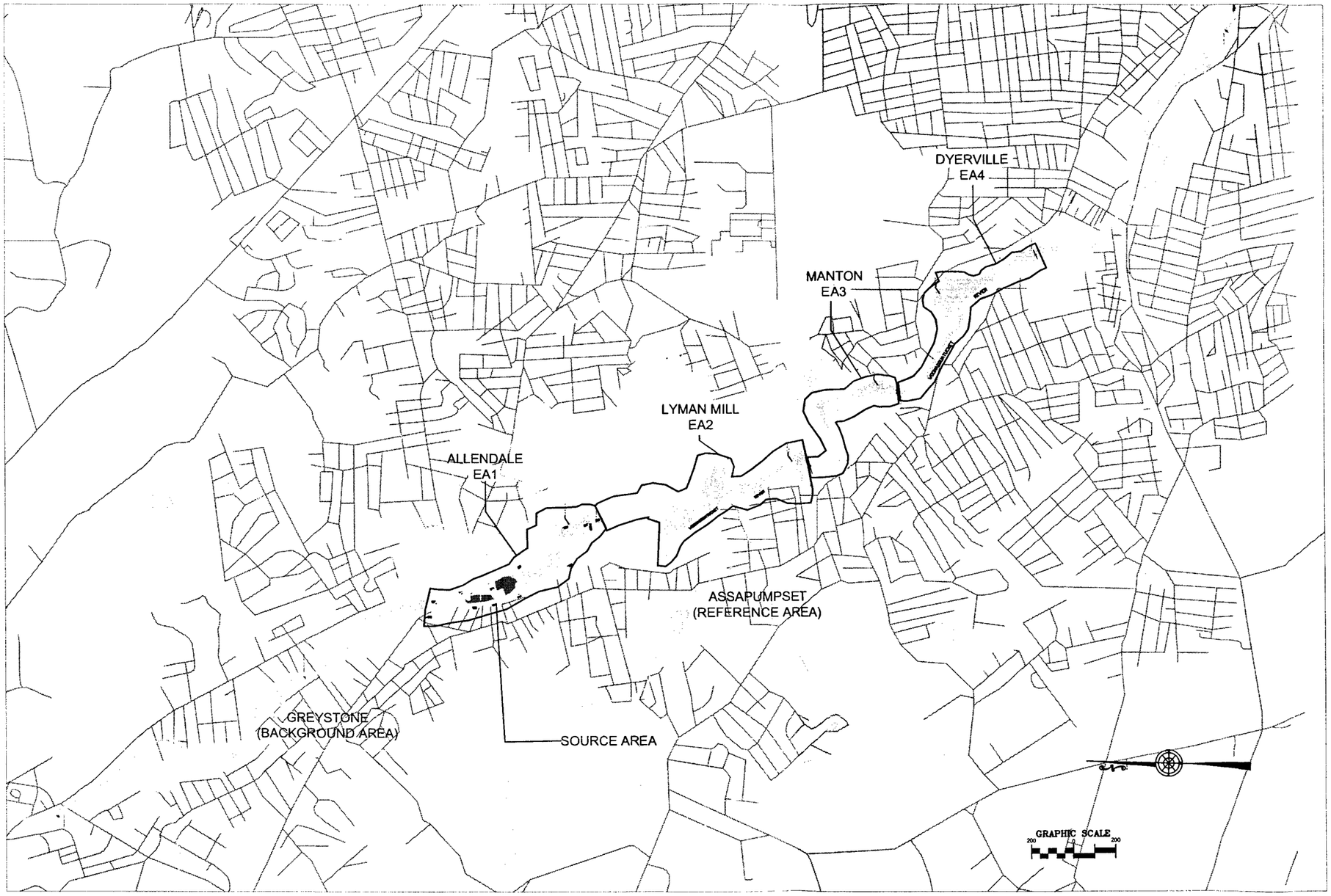
Site Locus Map
Baseline Ecological Risk Assessment

FIGURE: **1**

Centredale Manor Restoration Project Superfund
North Providence, Rhode Island

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BGF	51226-24	W2004027a	5/04	7/15/04

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LEGEND:

EA = EXPOSURE AREA

= SURFACE WATERS

= APPROXIMATE EXTENT OF EXPOSURE AREAS

* EA1, EA2, EA3, AND EA4 constitute the site.

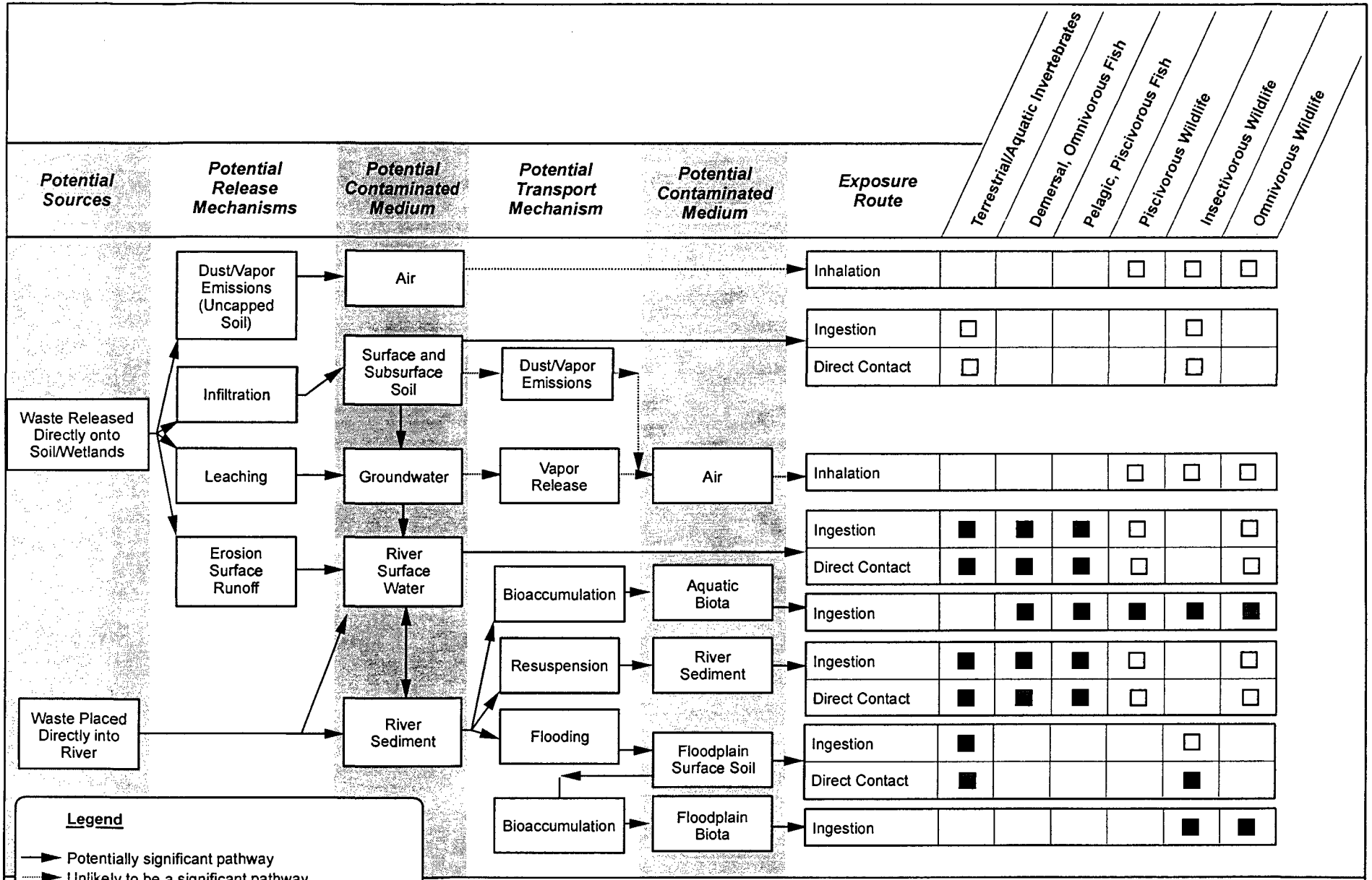
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FIGURE 2
SITE LAYOUT AND EXPOSURE AREAS
CENTREDALE MANOR RESTORATION PROJECT
SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

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Conceptual Site Model
Baseline Ecological Risk Assessment

Centredale Manor Restoration Project Superfund
North Providence, Rhode Island

FIGURE: **3**

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Legend:

- Pfo = Palustrine Forested
- Pem = Palustrine Emergent
- Pss = Palustrine Scrub/Shrub
- Pub = Palustrine Unconsolidated Bottom
- Rupub = Riverine Upper Perennial Unconsolidated Bottom
- Rlpub = Riverine Lower Perennial Unconsolidated Bottom
- Ruprb = Riverine Upper Perennial Rock Bottom
- Rlpub = Riverine Lower Perennial Unconsolidated Bottom
- Pfo/ss = Palustrine Forested/Scrub-Shrub
- Pem/ss = Palustrine Emergent/Scrub-Shrub
- Lliub = Lacustrine Littoral Unconsolidated Bottom
- Lliub/ab = Lacustrine Littoral Unconsolidated Bottom/Aquatic Bed
- Pub/ab = Palustrine Unconsolidated Bottom/Aquatic Bed
- Upland

ASSAPUMPSET BROOK

NORTH

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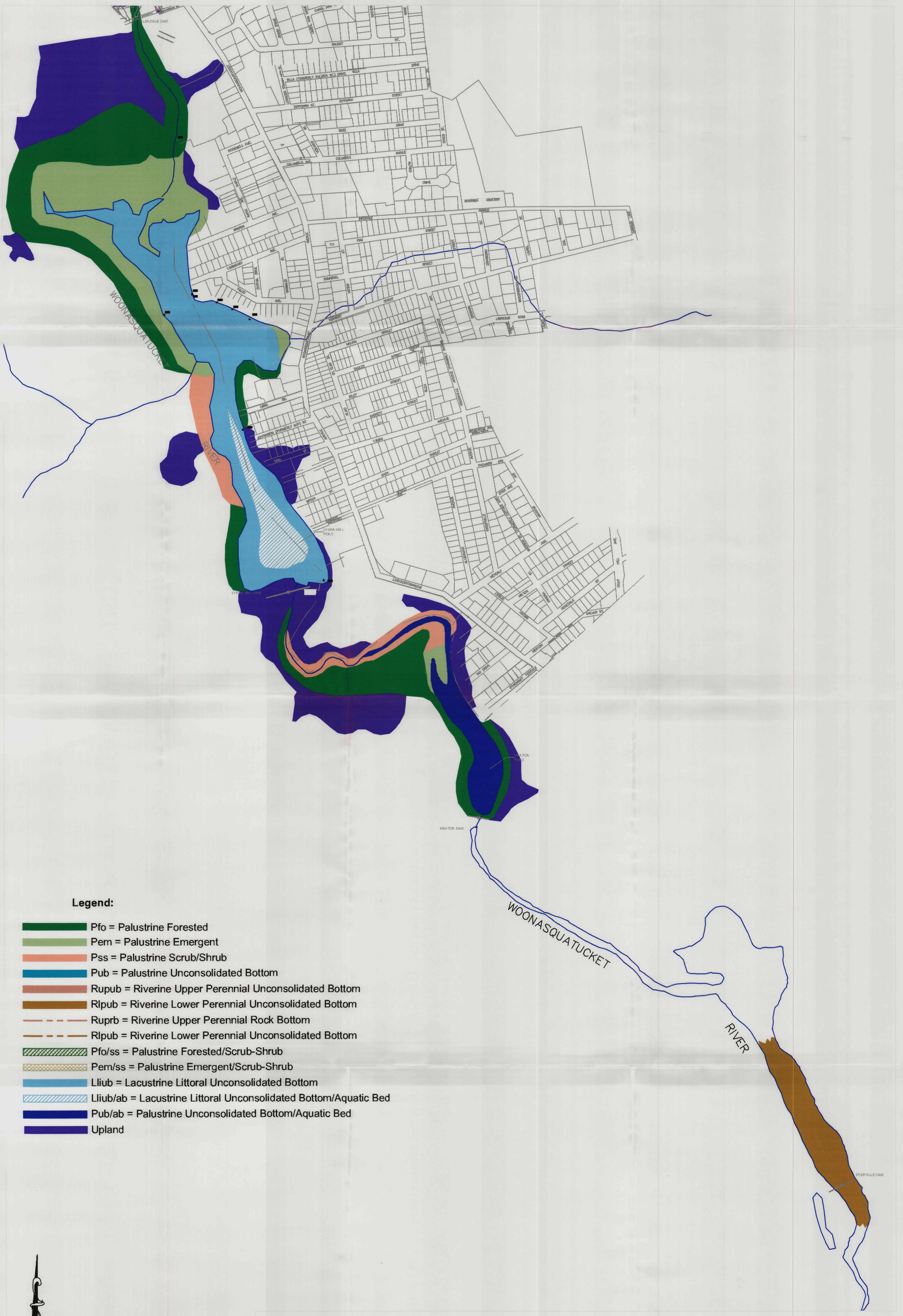
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FIGURE 4
ECOLOGICAL HABITATS ASSOCIATED WITH THE
WOONASQUATUCKET RIVER NORTH OF ALLENDALE DAM
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

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FIGURE 5
ECOLOGICAL HABITATS ASSOCIATED WITH THE
WOONASQUATUCKET RIVER SOUTH OF ALLENDALE DAM
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

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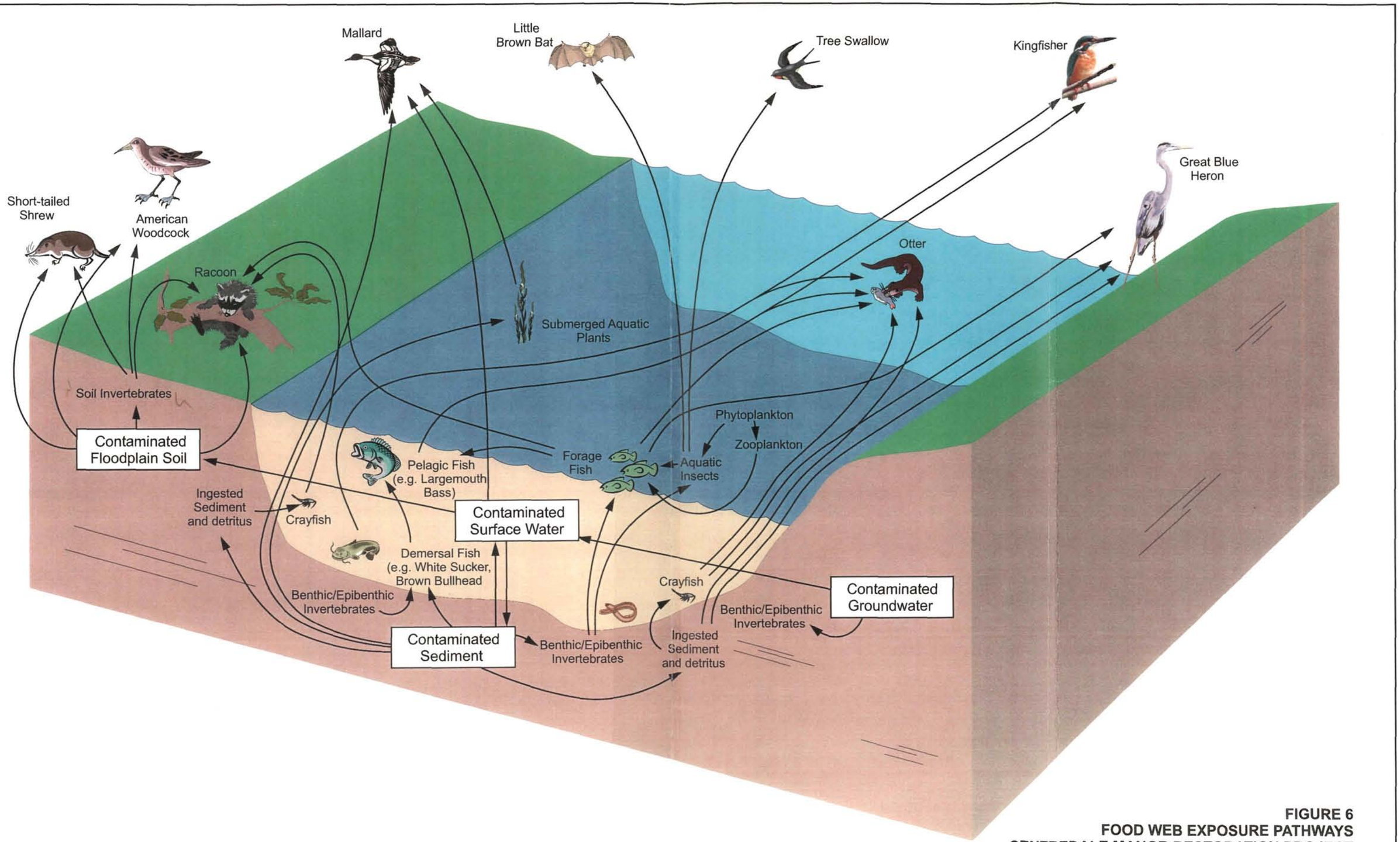


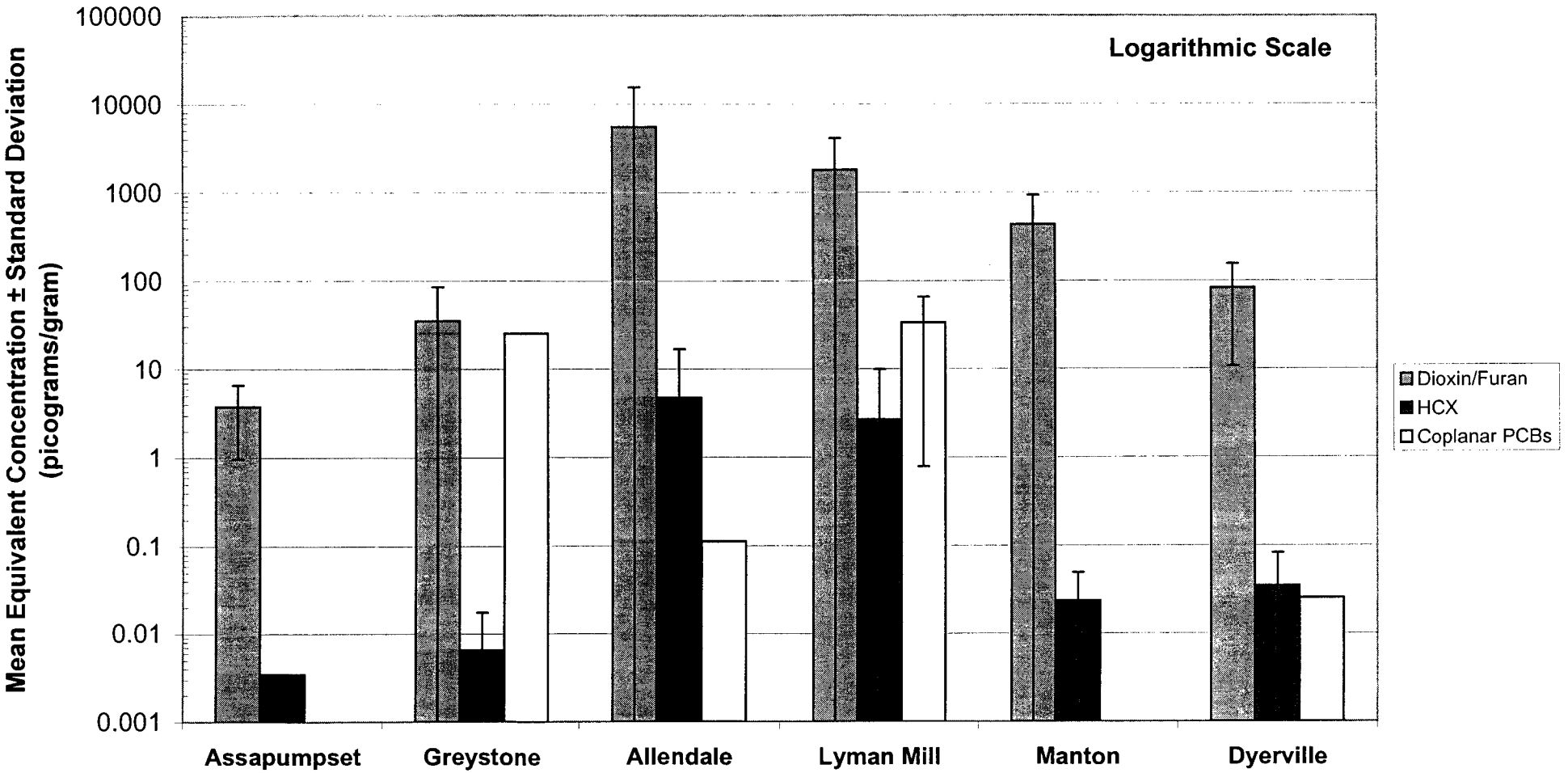
FIGURE 6
FOOD WEB EXPOSURE PATHWAYS
CENTREDALE MANOR RESTORATION PROJECT
SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

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Figure 7
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Sediment

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Picograms/gram is equivalent to parts per trillion.

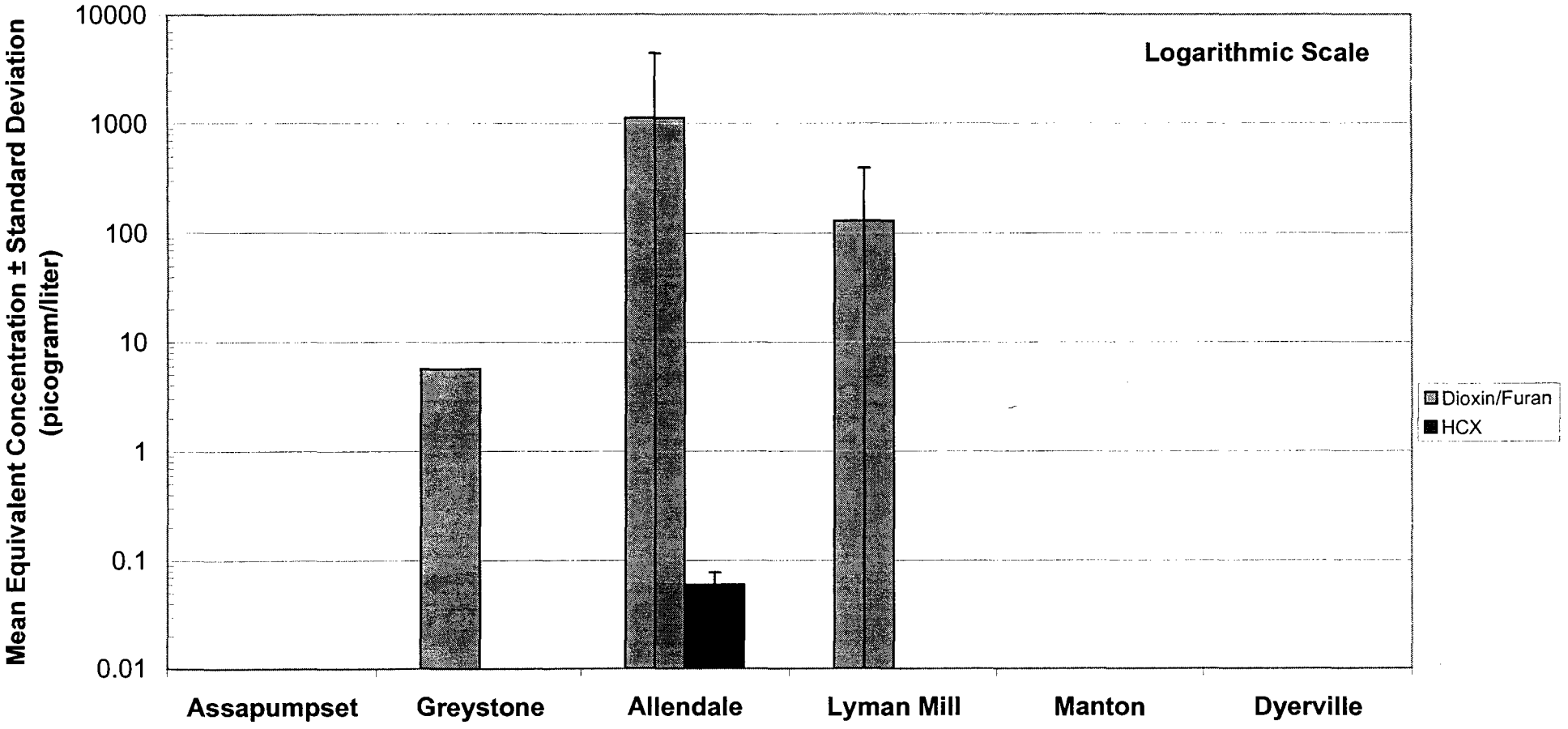
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Figure 8
Dioxin/Furan and HCX Equivalent Concentrations in Surface Water

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



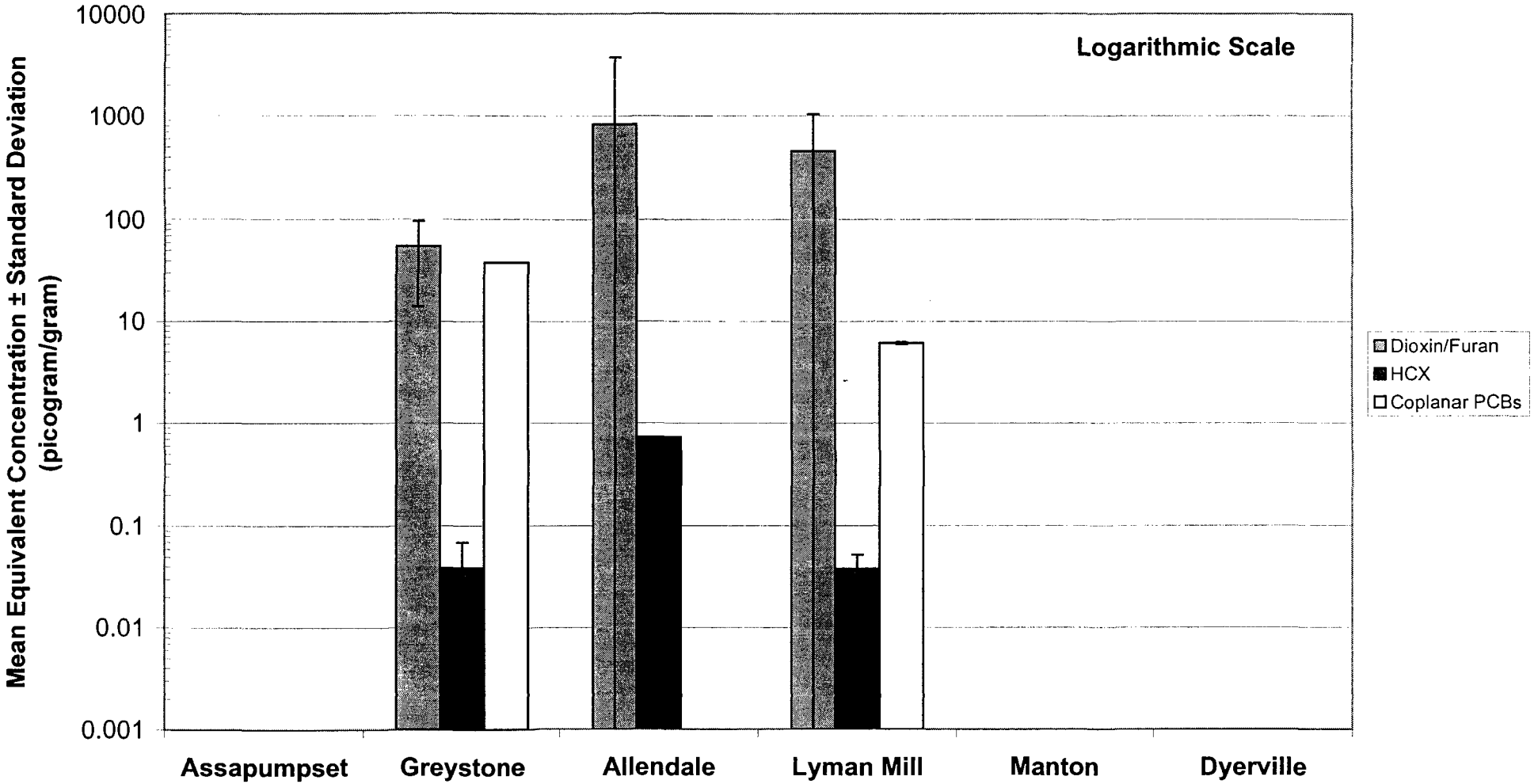
Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Coplanar PCB analysis was not conducted for surface water.
Picograms per liter is equivalent to parts per quadrillion.

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Figure 9
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Floodplain and Bank Soil

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI



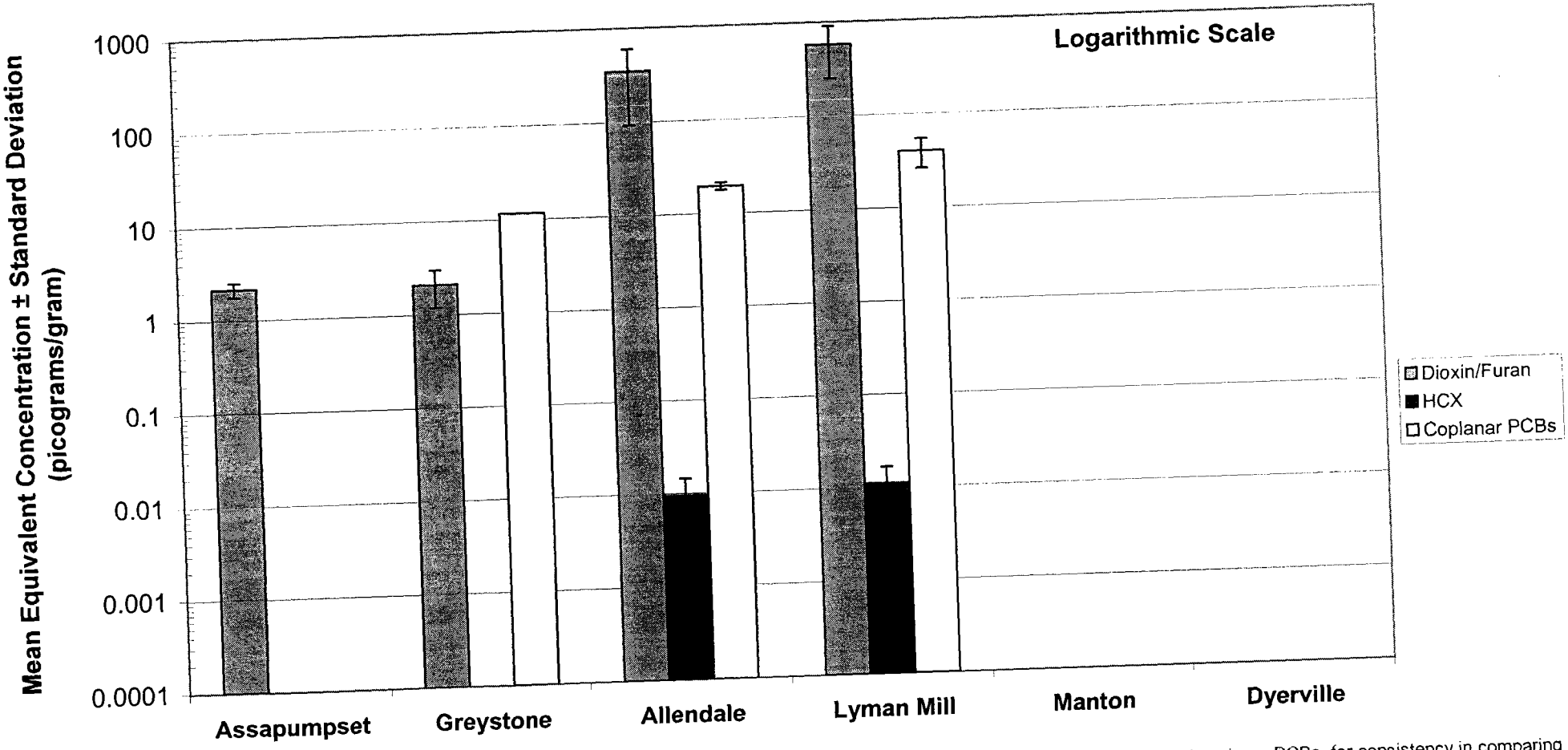
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Picogram/gram is equivalent to parts per trillion.

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Figure 10
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in White Sucker

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Picograms/gram is equivalent to parts per trillion.

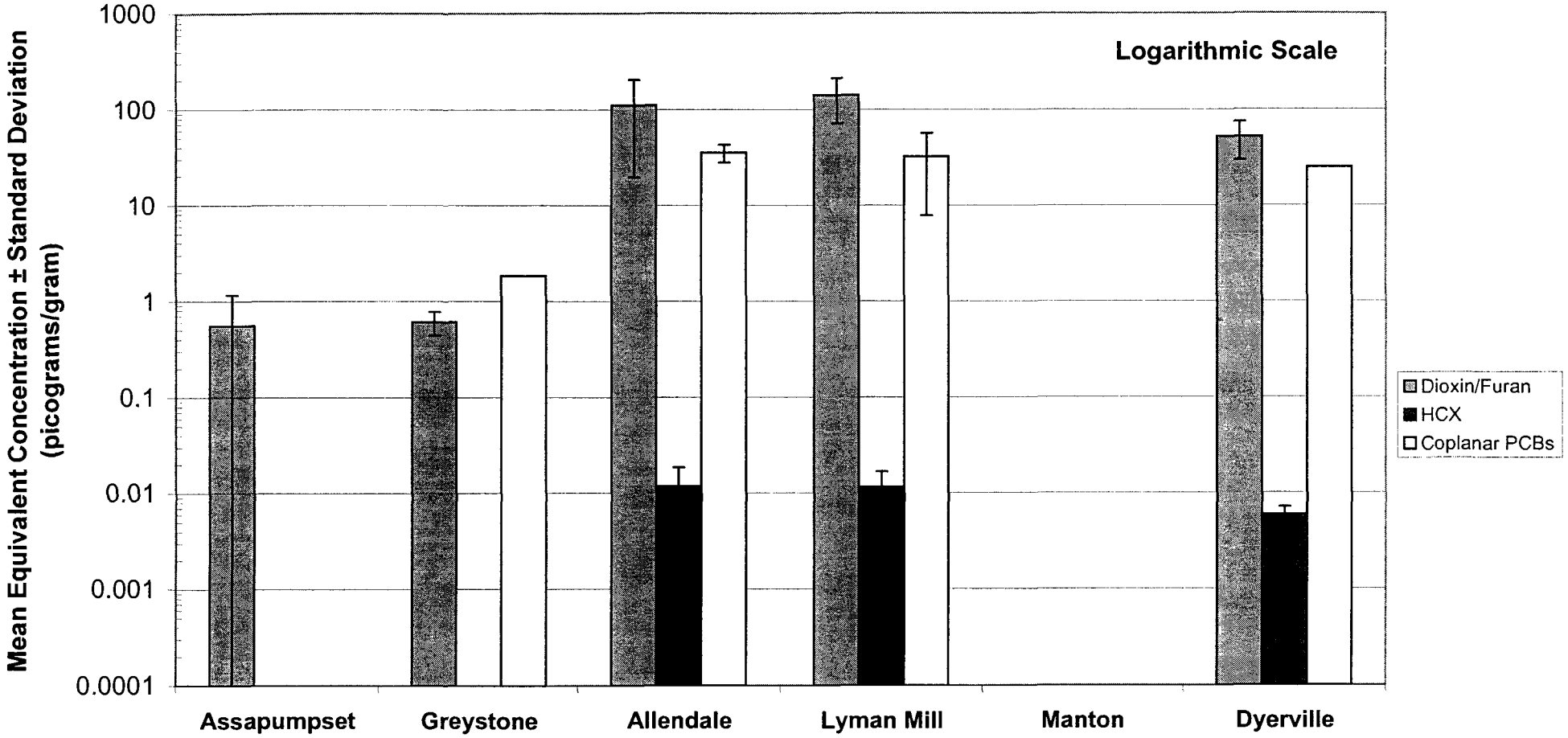
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Figure 11
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in American Eel

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

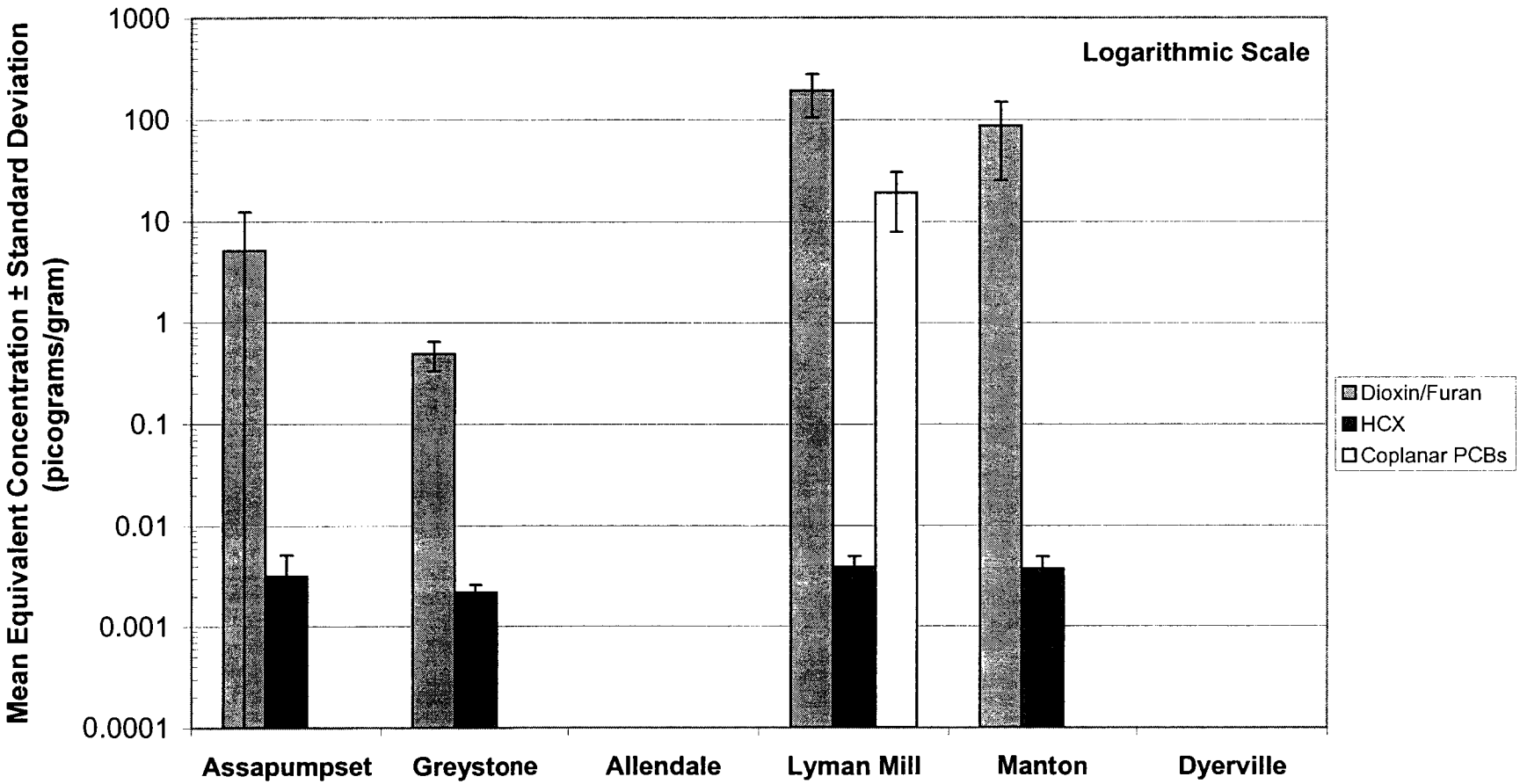
Picograms/gram is equivalent to parts per trillion.

MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\EelEel, Dioxin Chart 1

8/5/2004

Figure 12
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Largemouth Bass

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI

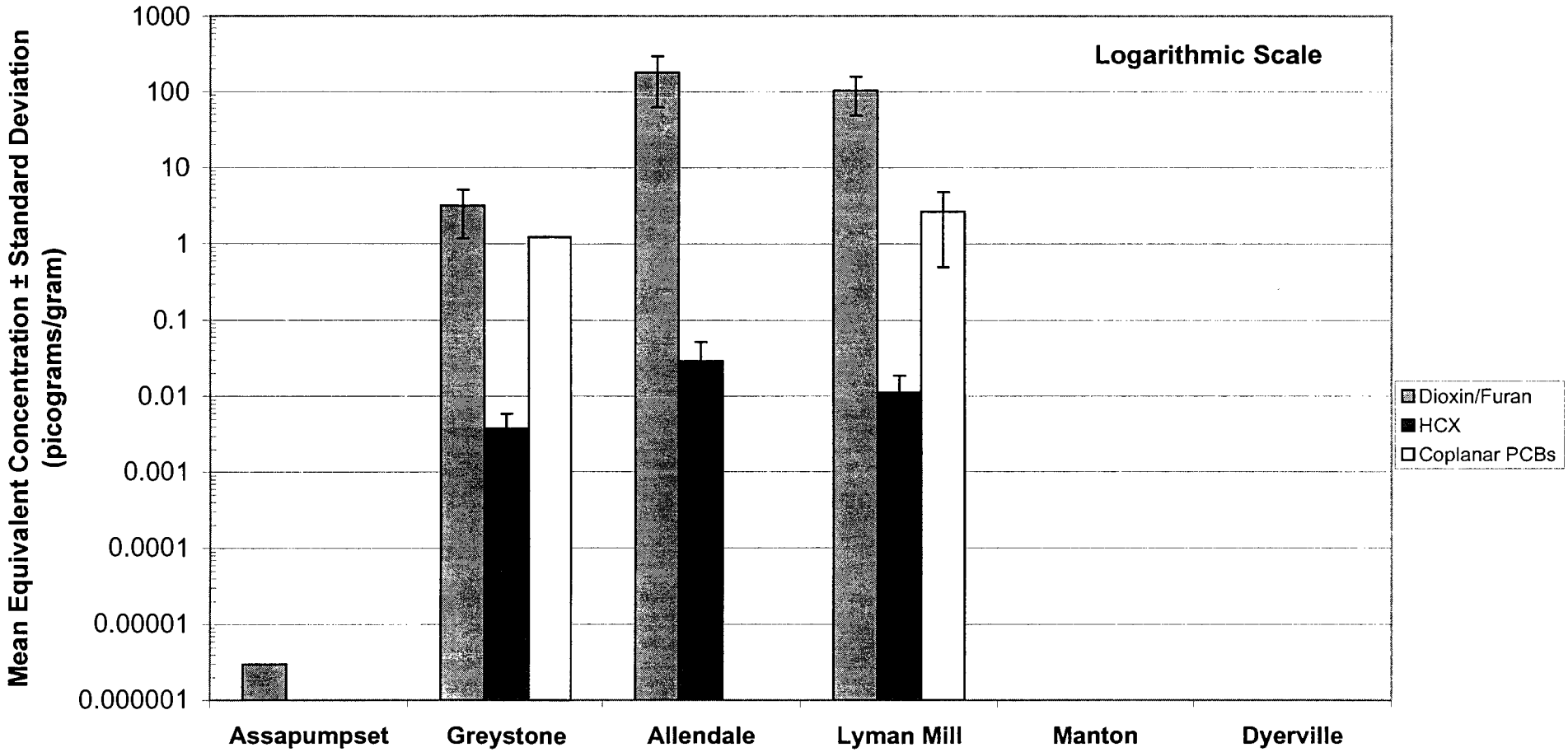


Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Reconstituted whole body data for Largemouth Bass used in developing histograms.
Picograms/gram is equivalent to parts per trillion.

Figure 13
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Crayfish

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

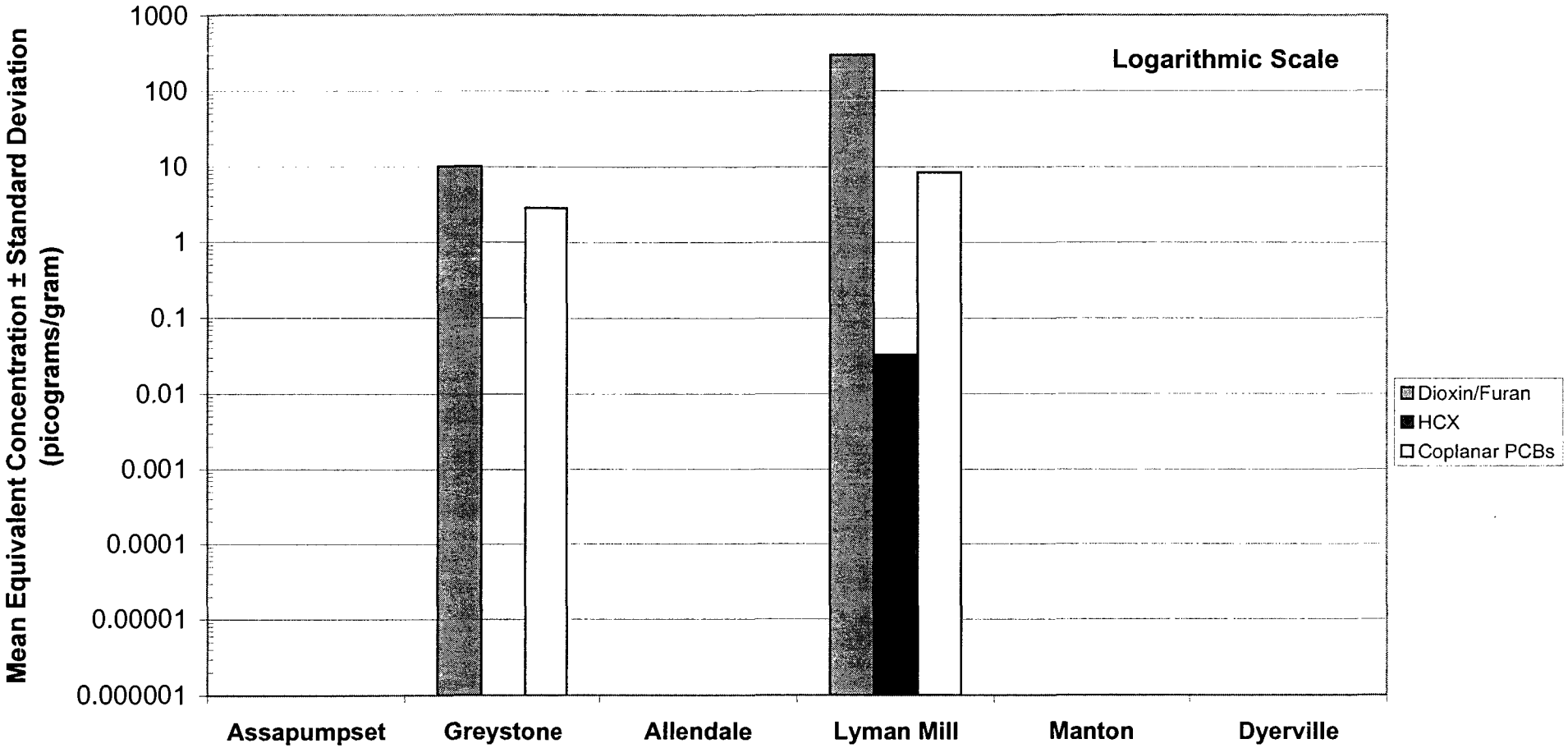
Picograms/gram is equivalent to parts per trillion.

MACTEC Engineering and Consulting, Inc.
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Figure 14
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Emerging Insects

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, RI

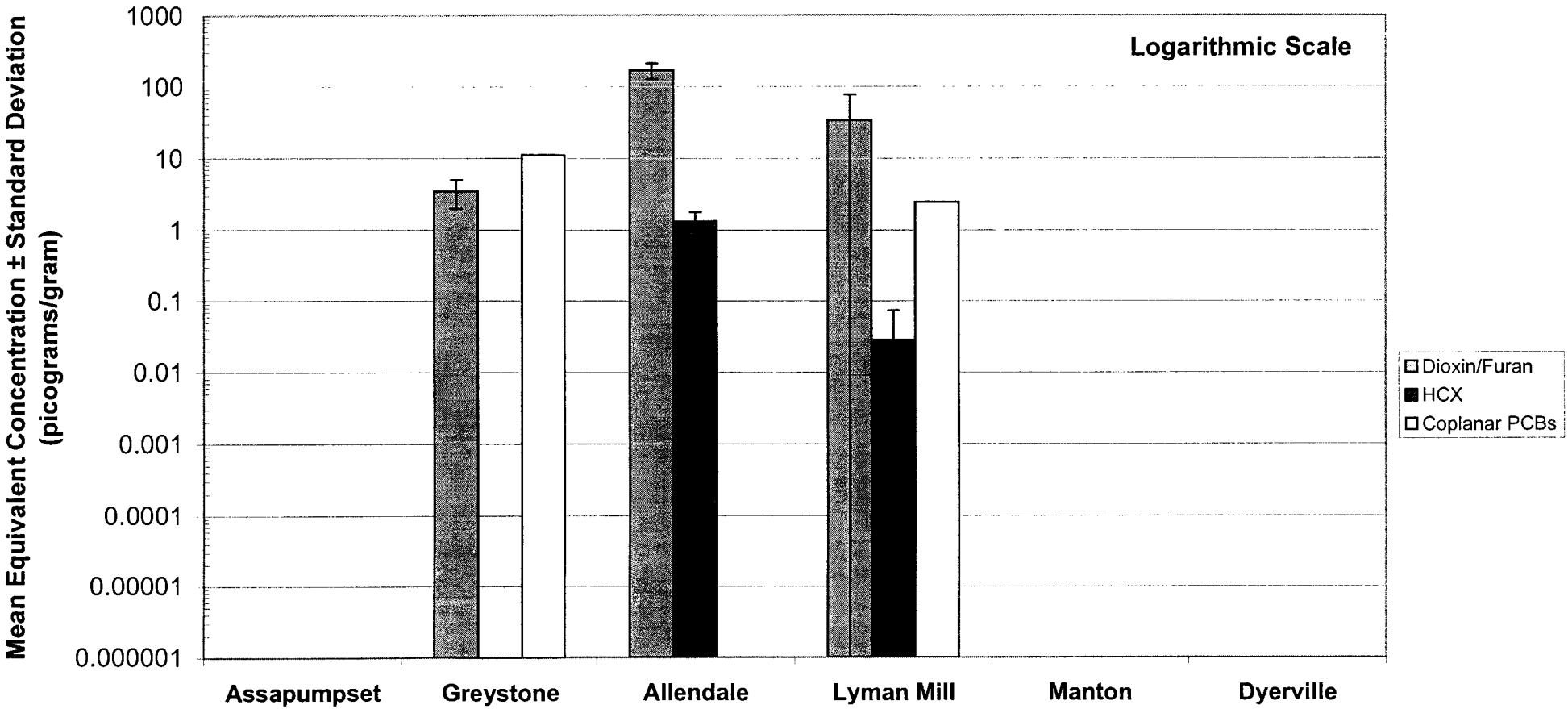


Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Picograms/gram is equivalent to parts per trillion.

Figure 15
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Earthworms

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Mammalian toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-mammalian receptors.

Picograms/gram is equivalent to parts per trillion.

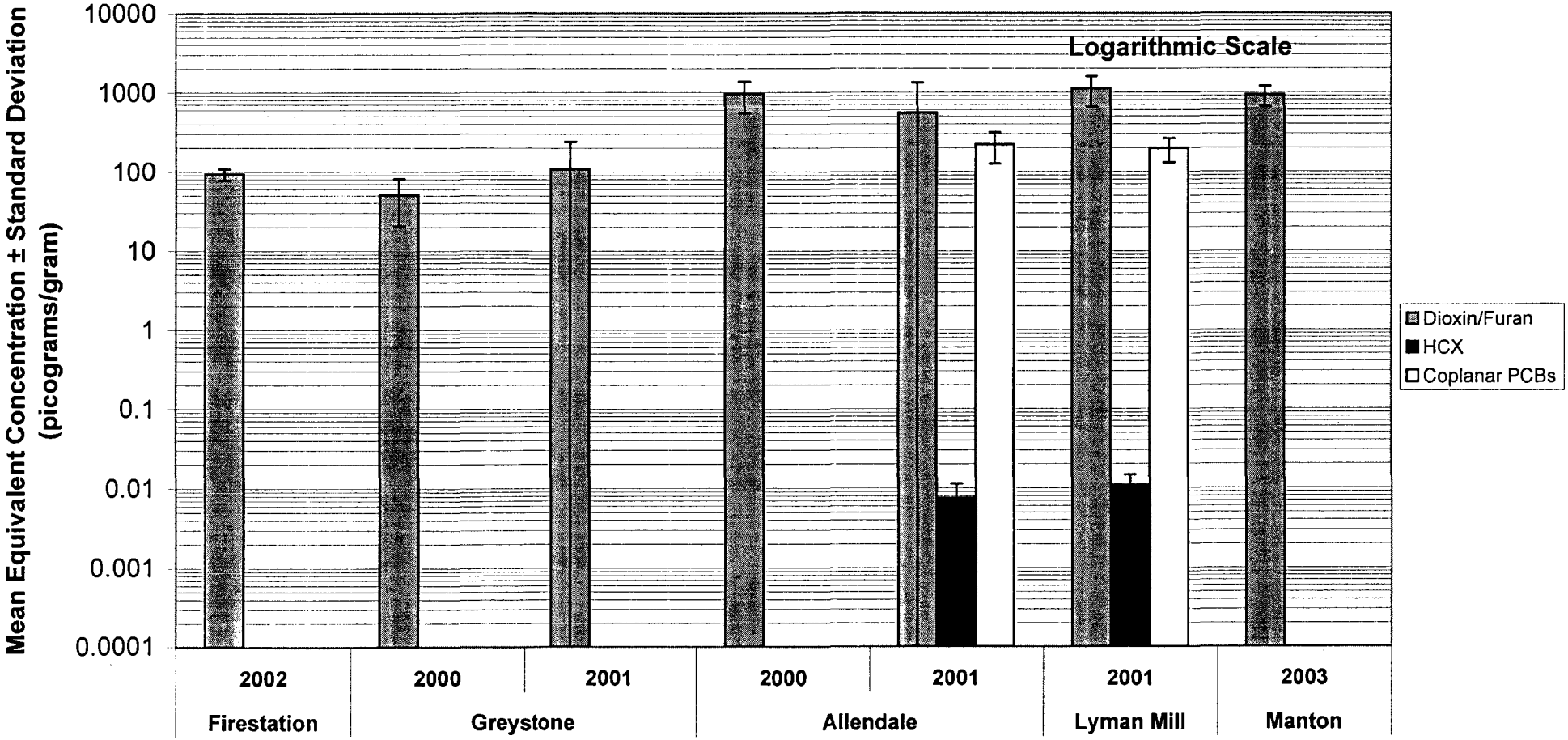
MACTEC Engineering and Consulting, Inc.
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Figure 16
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Tree Swallow Egg

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Bird toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-bird receptors.

Picograms/gram is equivalent to parts per trillion.

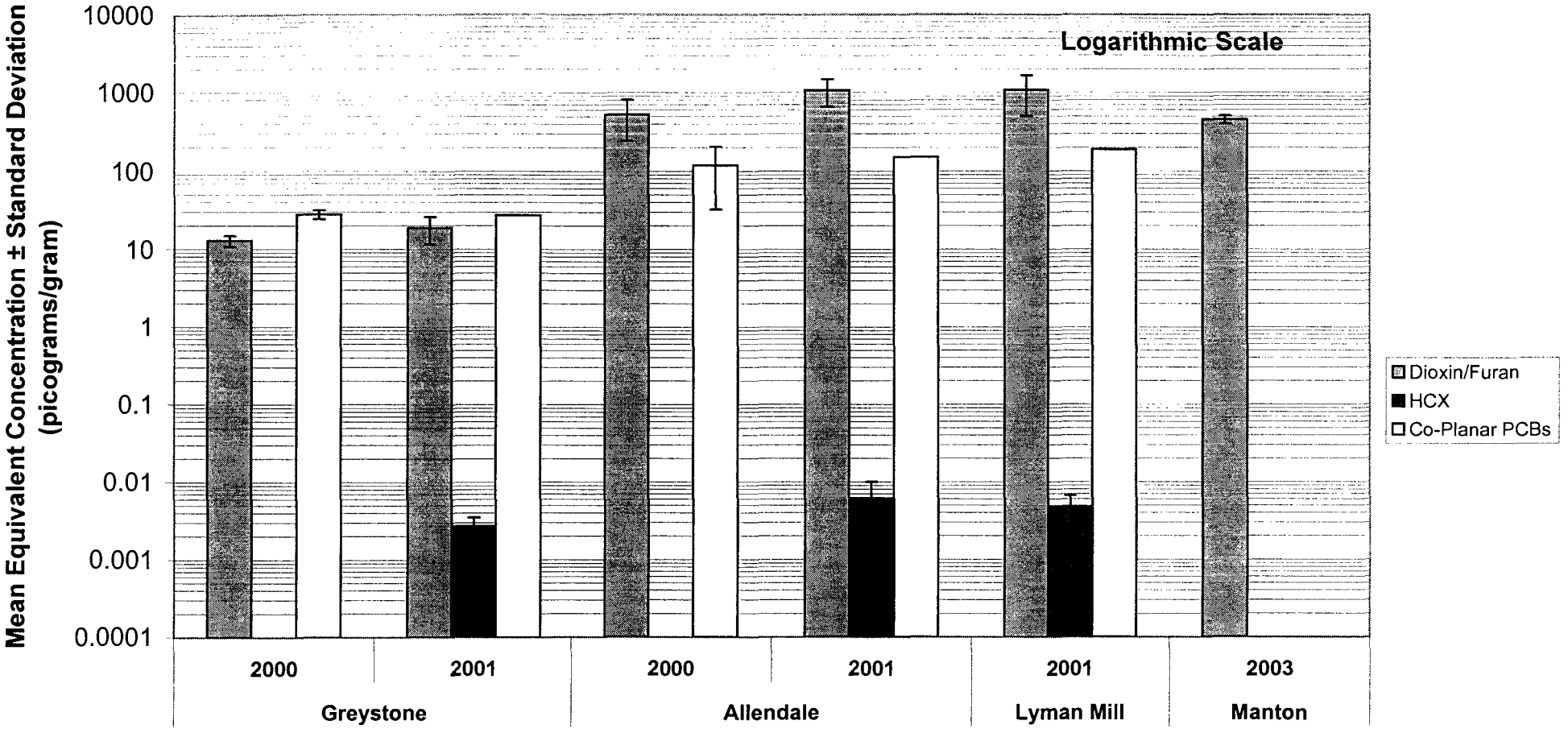
MACTEC Engineering and Consulting, Inc.
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P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\TreeSwallowEgg(NoSwitch).xls, Dioxin Chart 1

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Figure 17
Dioxin/Furan, HCX, and Coplanar PCB Equivalent Concentrations in Tree Swallow Nestling

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Bird toxicity equivalence factors (TEFs) per Van den Berg et al. 1998 were used to calculate equivalent concentrations for dioxins, furans, and coplanar PCBs, for consistency in comparing nature and extent of contamination only. Hexachloroxanthene (HCX) equivalent concentrations were calculated using a TEF of 0.0002. These equivalent concentrations are not intended to be representative of ecological risks for non-bird receptors.

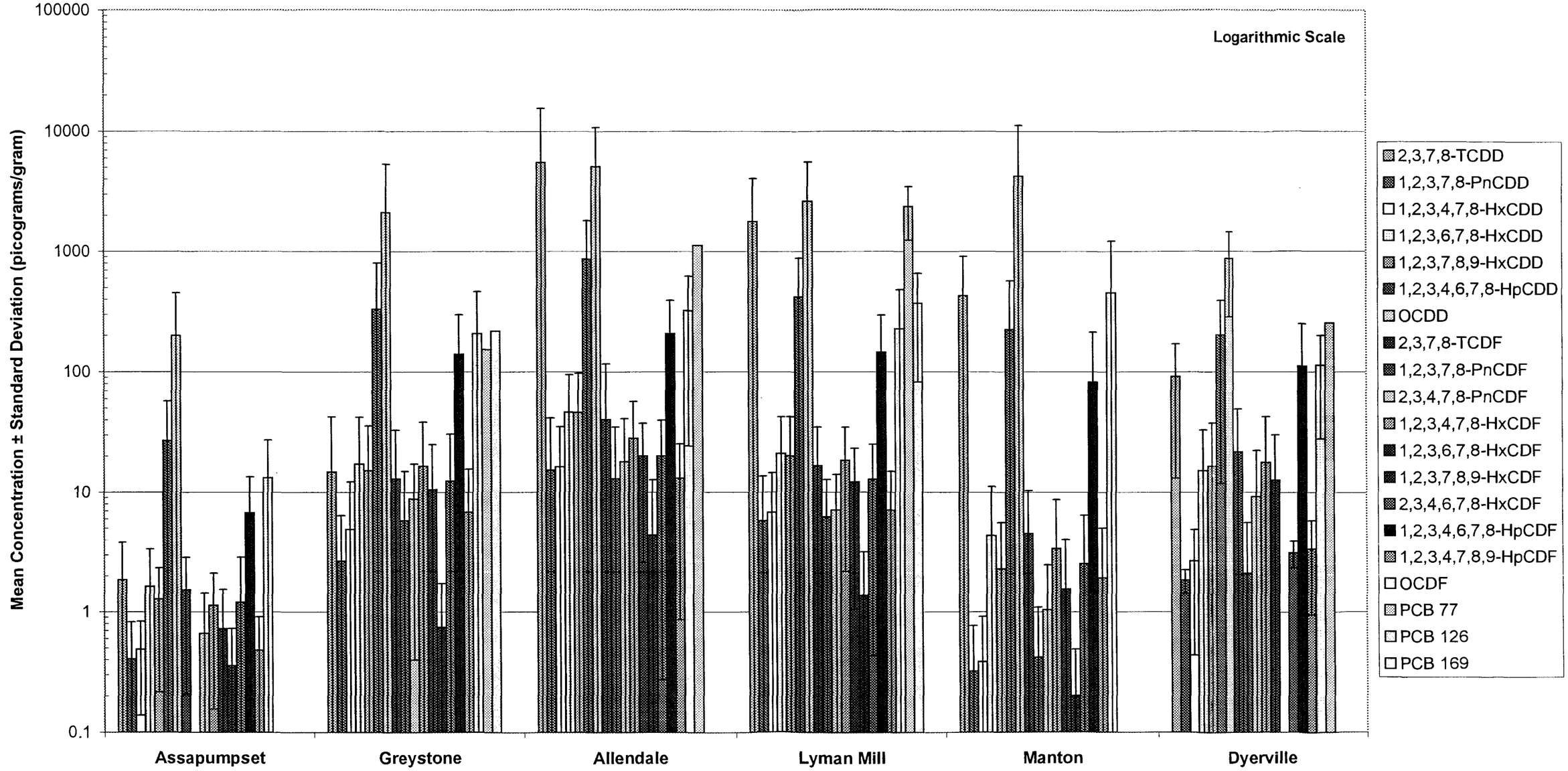
Picograms/gram is equivalent to parts per trillion.

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Figure 18
Select Dioxin and PCB Congener Concentrations in Sediment

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picograms/gram is equivalent to parts per trillion.

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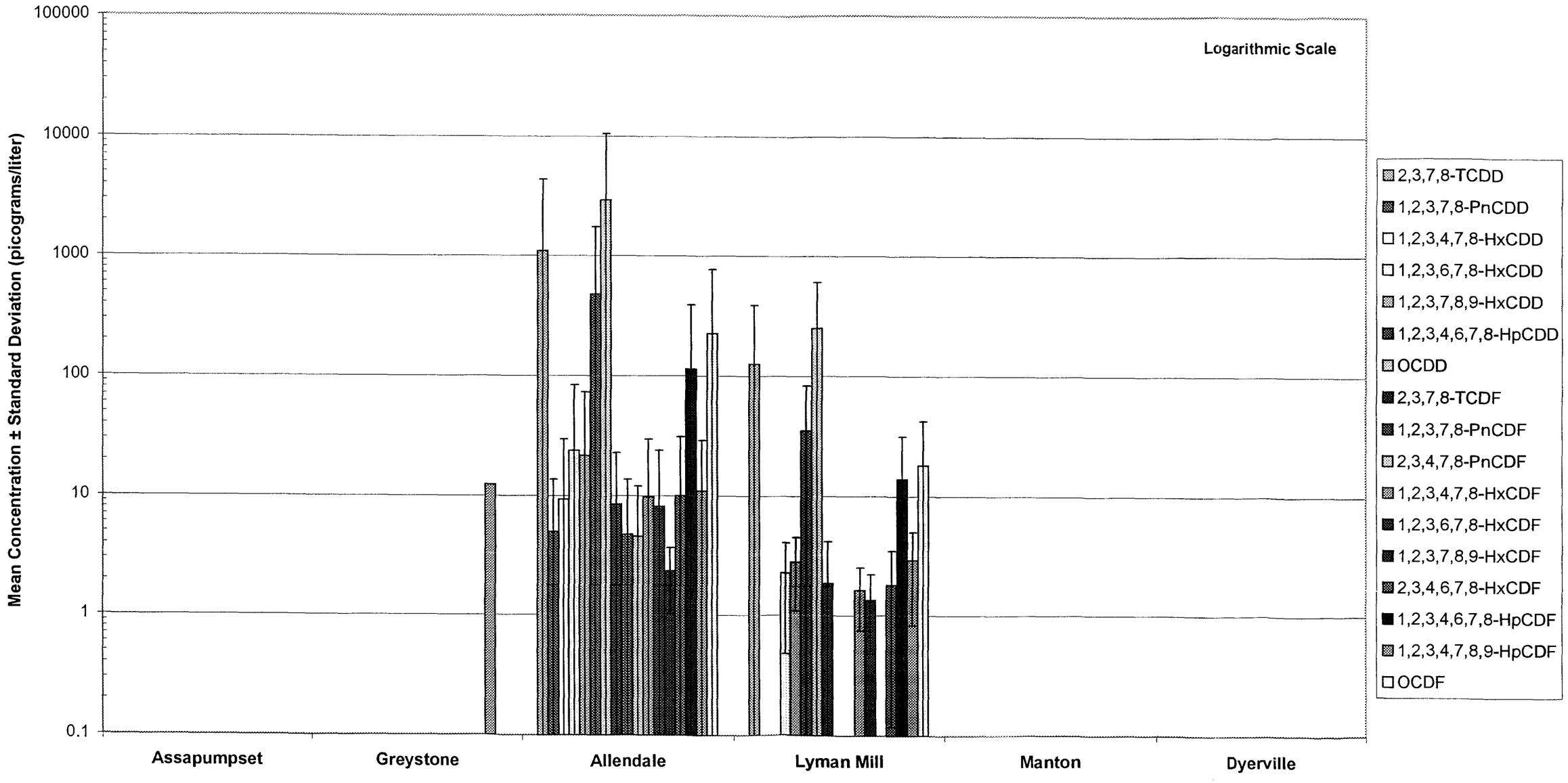
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Figure 19
Select Dioxin Concentrations in Surface Water

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

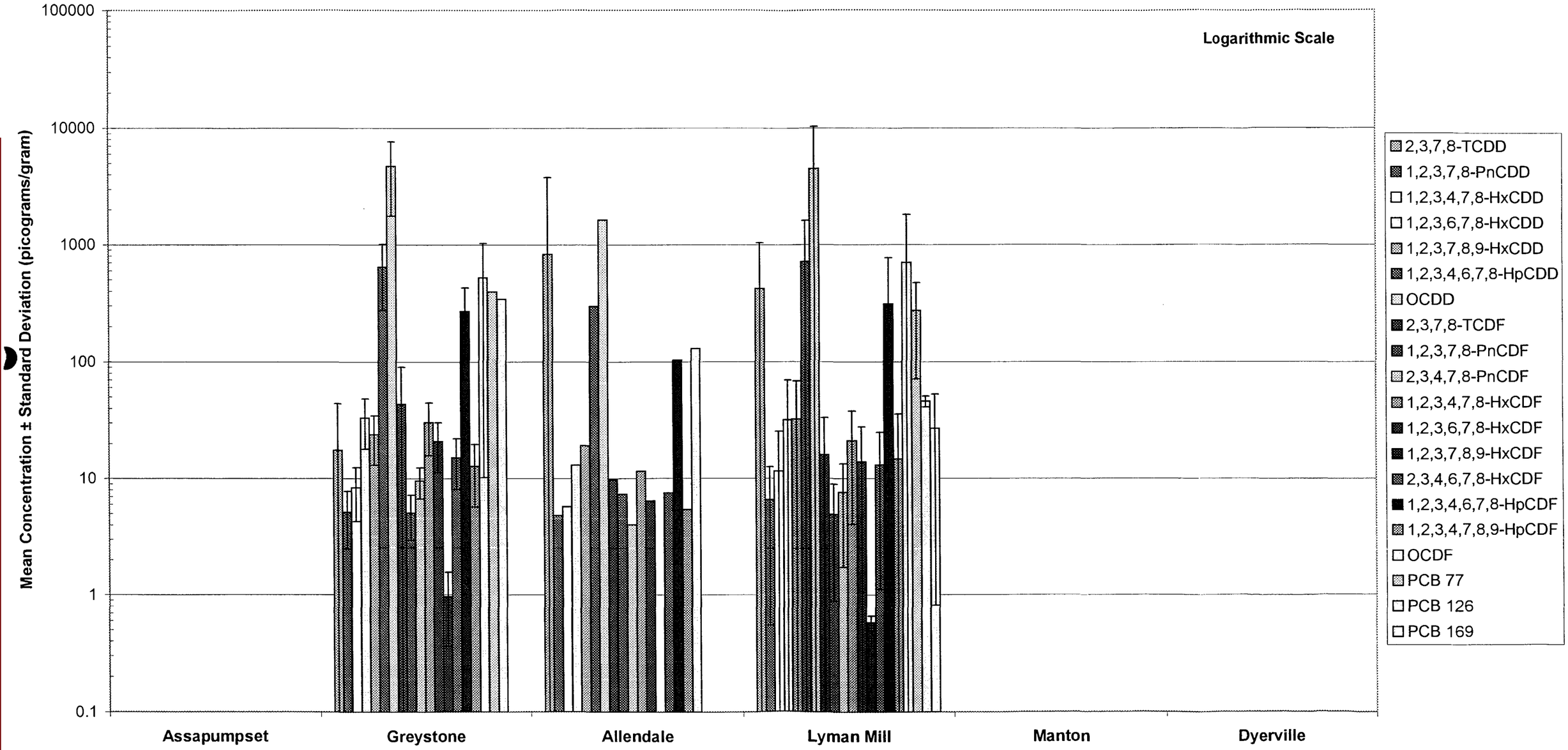


Coplanar PCB analysis was not conducted for surface water.
Picograms per liter is equivalent to parts per quadrillion.

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Figure 20
Select Dioxin and PCB Congener Concentrations in Floodplain and Bank Soil

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

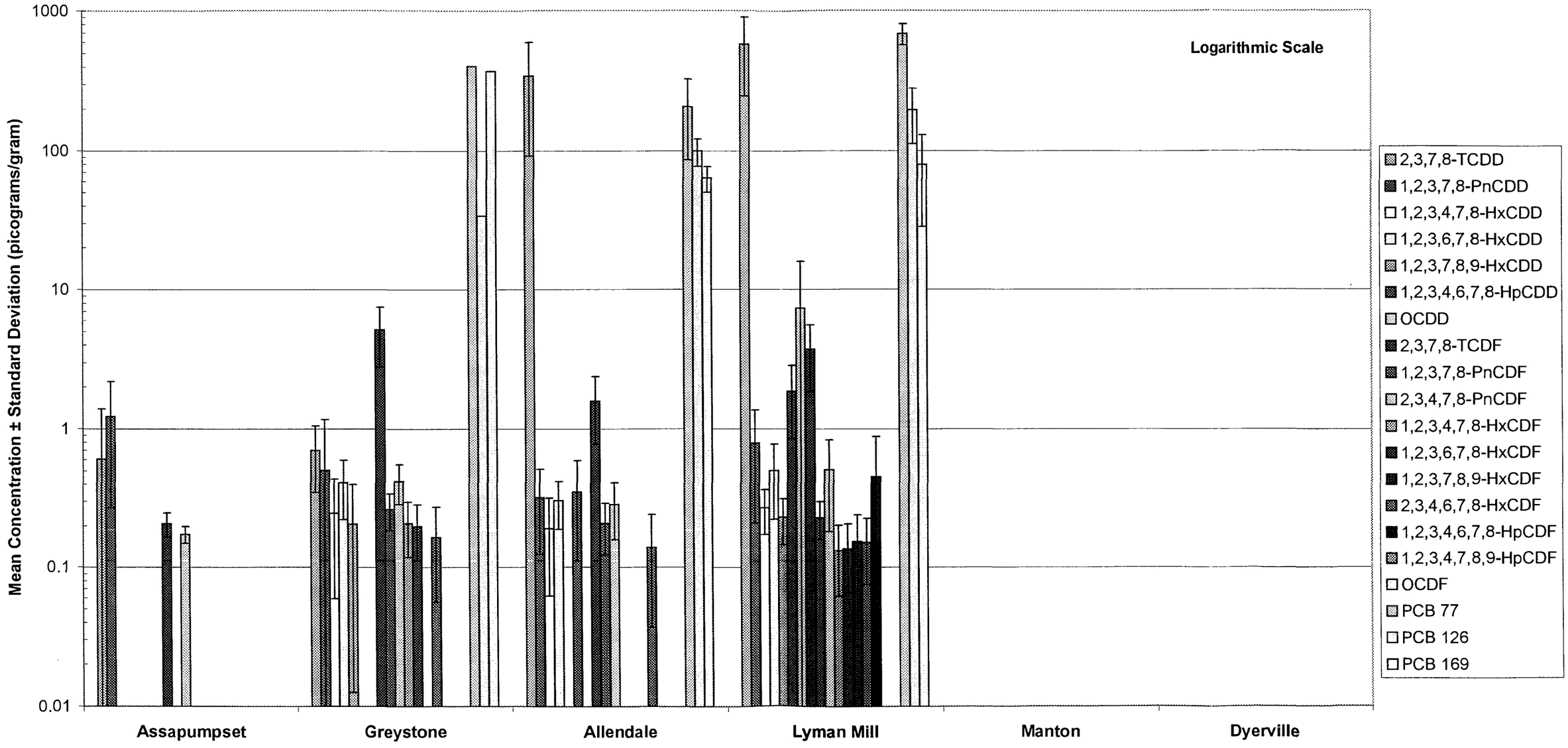
MACTEC Engineering and Consulting, Inc.
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Figure 21
Select Dioxin and PCB Congener Concentrations in White Sucker

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



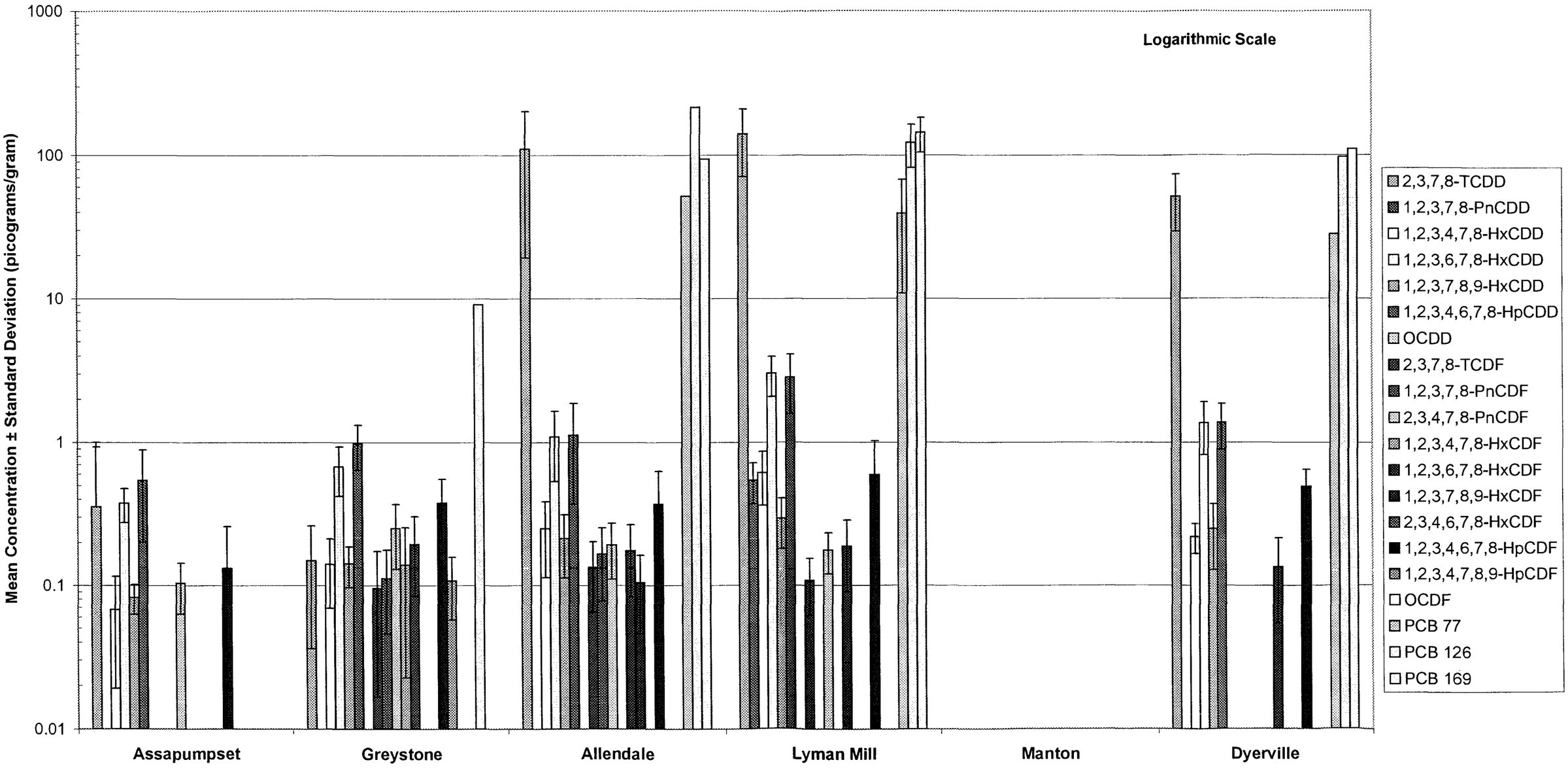
Note: Brown Bullhead were used as a surrogate for White Sucker in Assapumpset Brook
Picograms/gram is equivalent to parts per trillion.

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Figure 22
Select Dioxin and PCB Congener Concentrations in American Eel

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

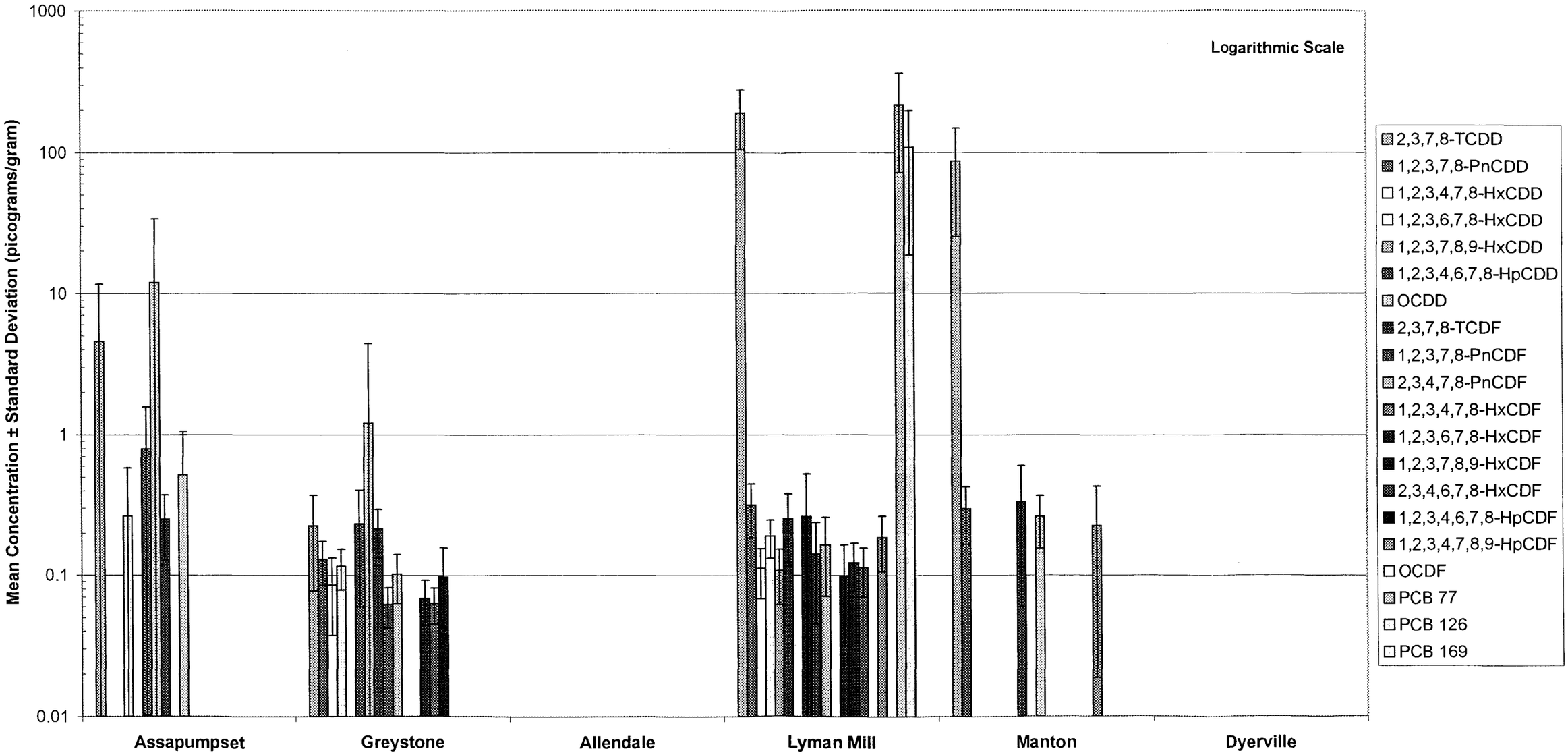
MACTEC Engineering and Consulting, Inc.
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Figure 23
Select Dioxin and PCB Congener Concentrations in Largemouth Bass

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Reconstituted whole body data for Largemouth Bass used in developing histograms.
Picogram/gram is equivalent to parts per trillion.

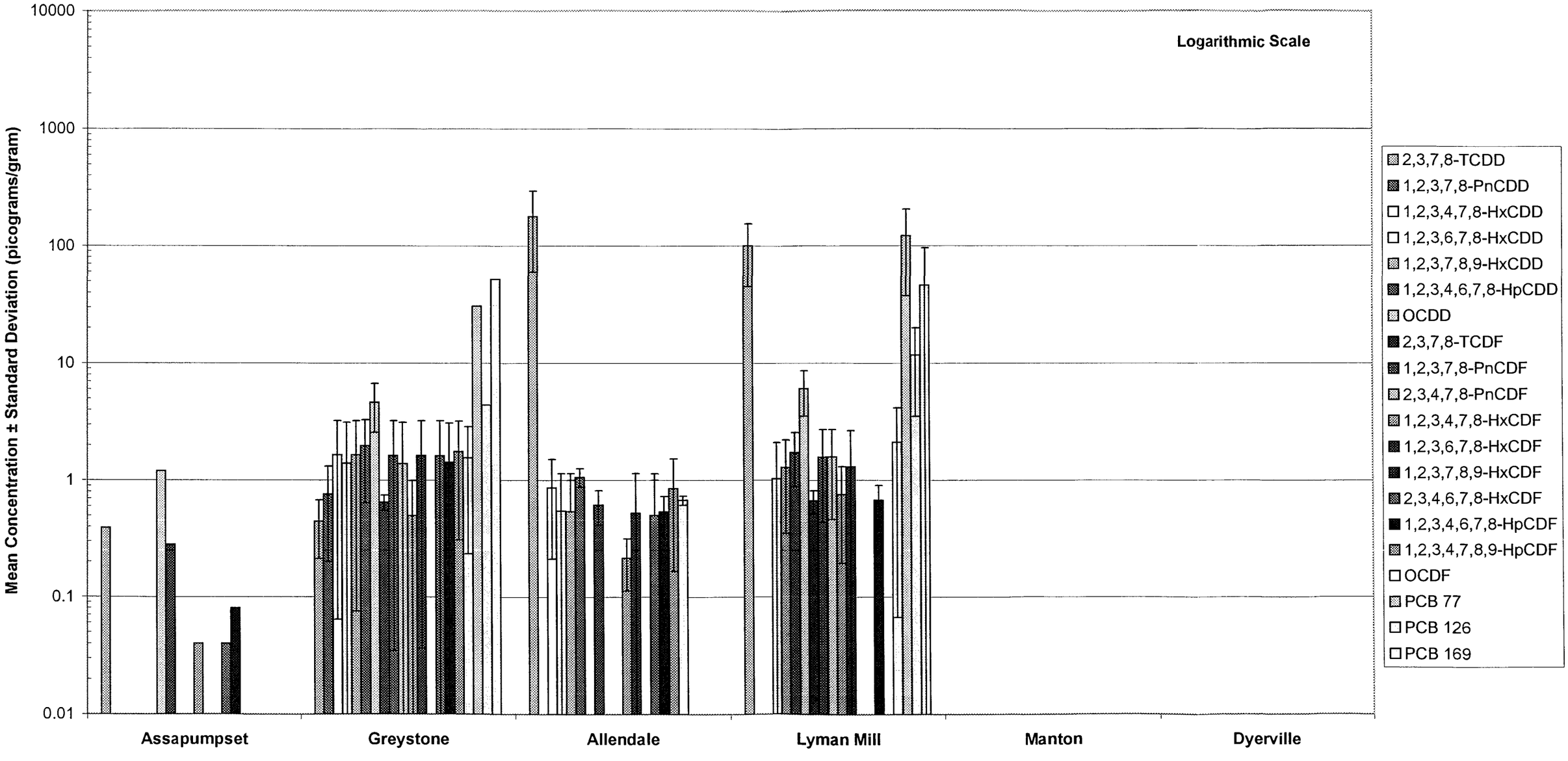
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Figure 24
Select Dioxin and PCB Congener Concentrations in Crayfish

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



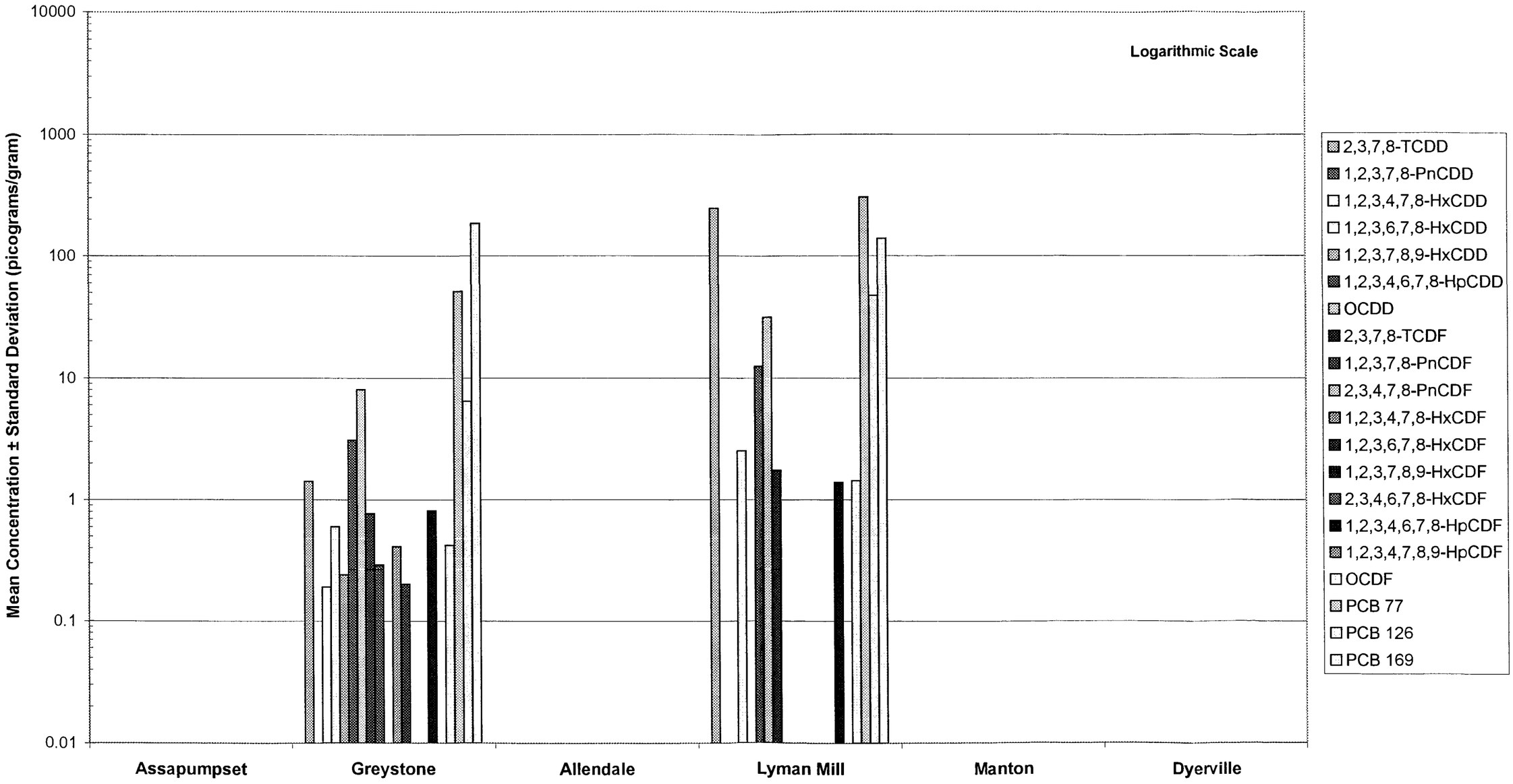
Picogram/gram is equivalent to parts per trillion.

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Figure 25
Select Dioxin and PCB Congener Concentrations in Emerging Insects

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

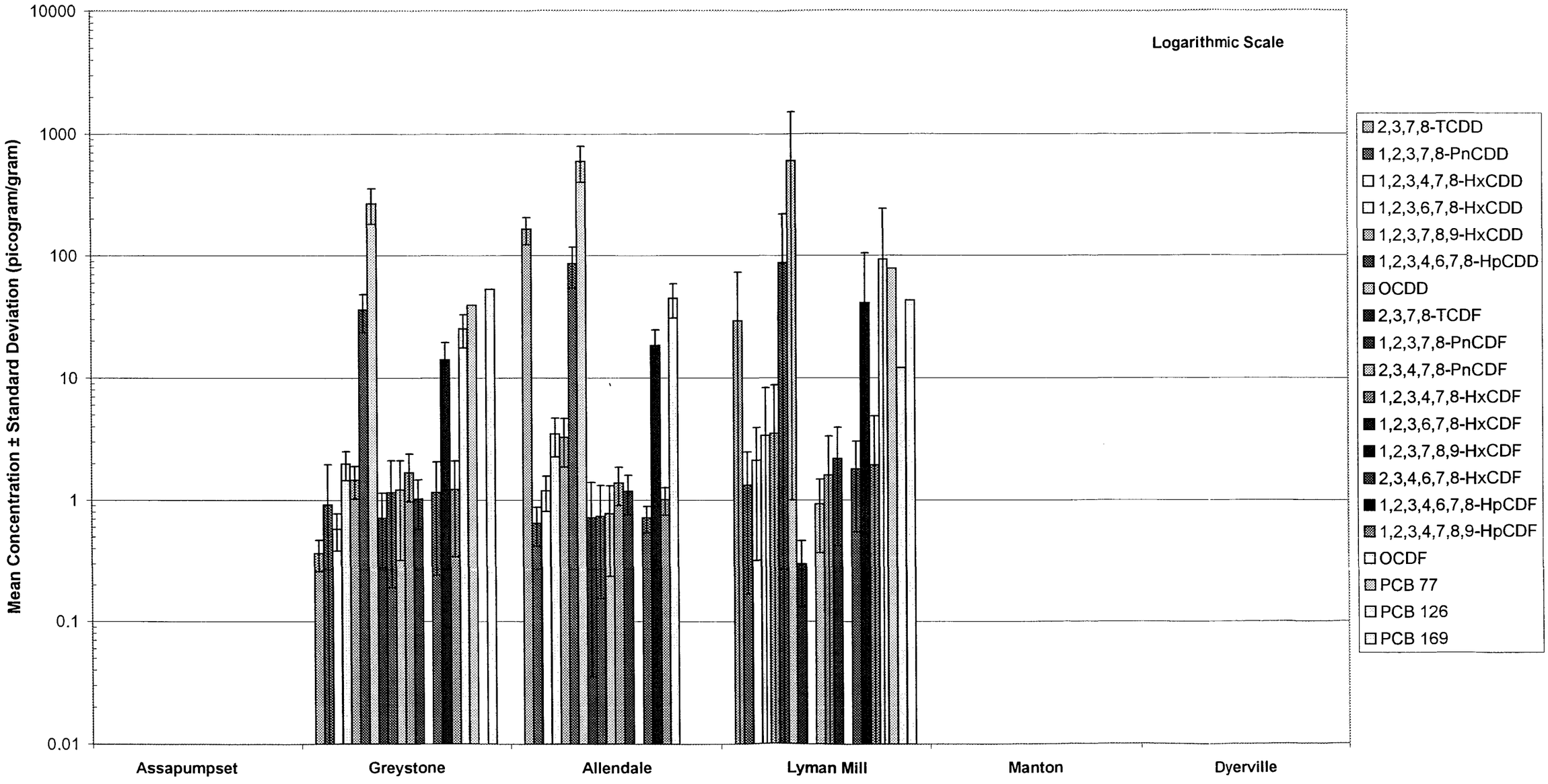
MACTEC Engineering and Consulting, Inc.
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Figure 26
Select Dioxin and PCB Congener Concentrations in Earthworms

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

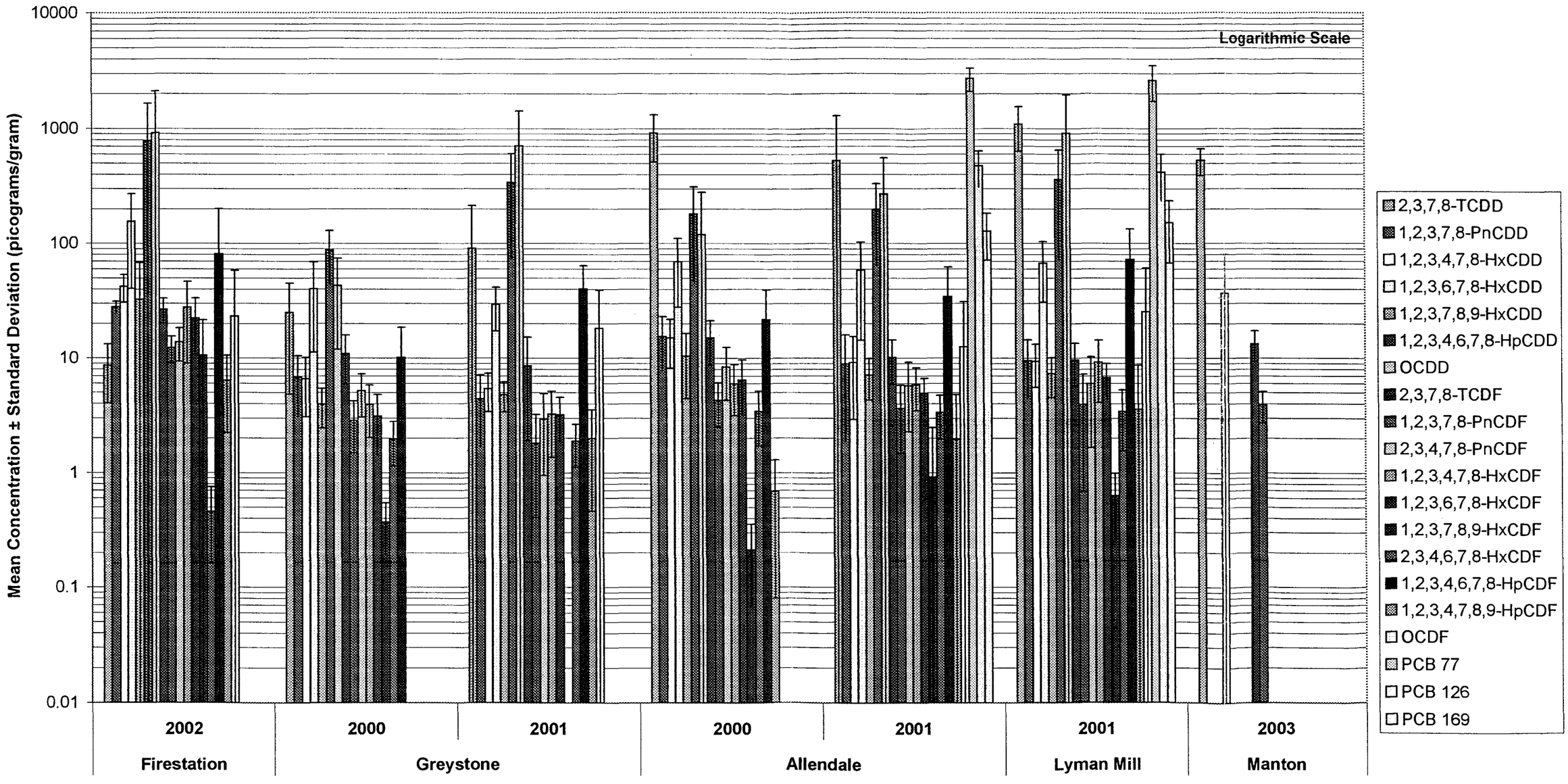
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Figure 27
Select Dioxin and PCB Congener Concentrations in Tree Swallow Egg

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Picogram/gram is equivalent to parts per trillion.

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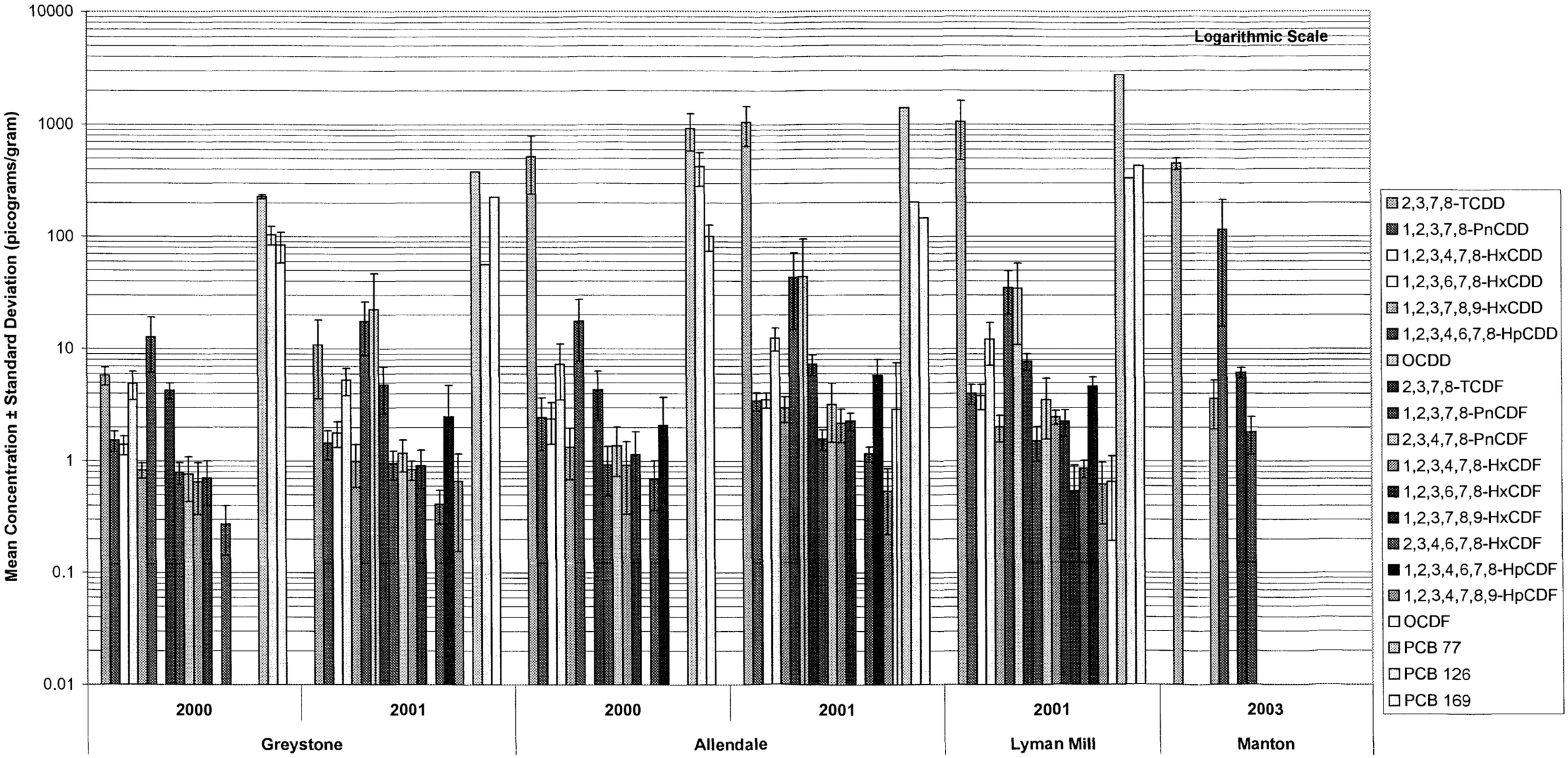
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Figure 28
Select Dioxin and PCB Congener Concentrations in Tree Swallow Nestling

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



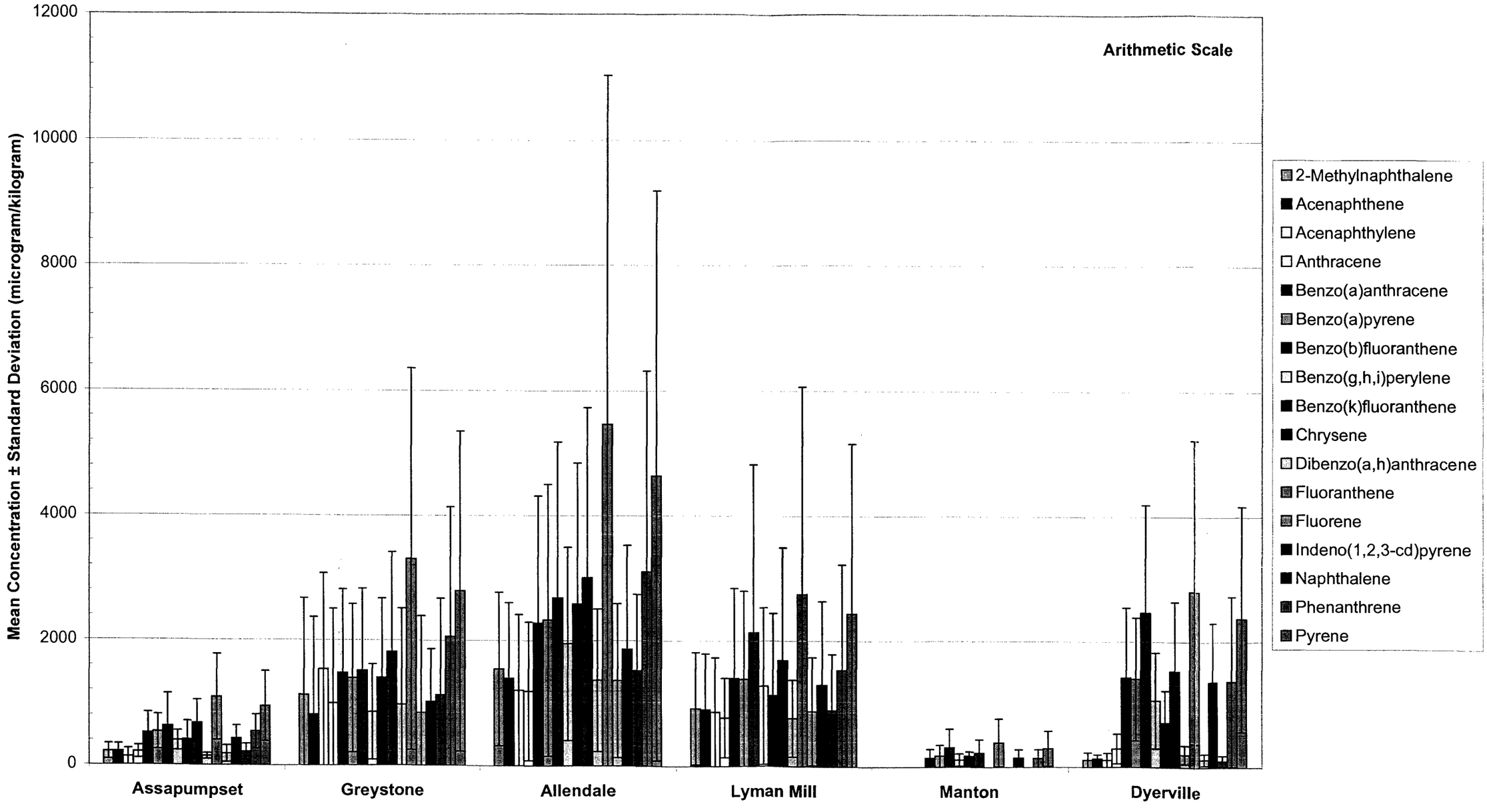
Picogram/gram is equivalent to parts per trillion.

MACTEC Engineering and Consulting, Inc.
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8/5/2004

Figure 29
Select PAH Concentrations in Sediment

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



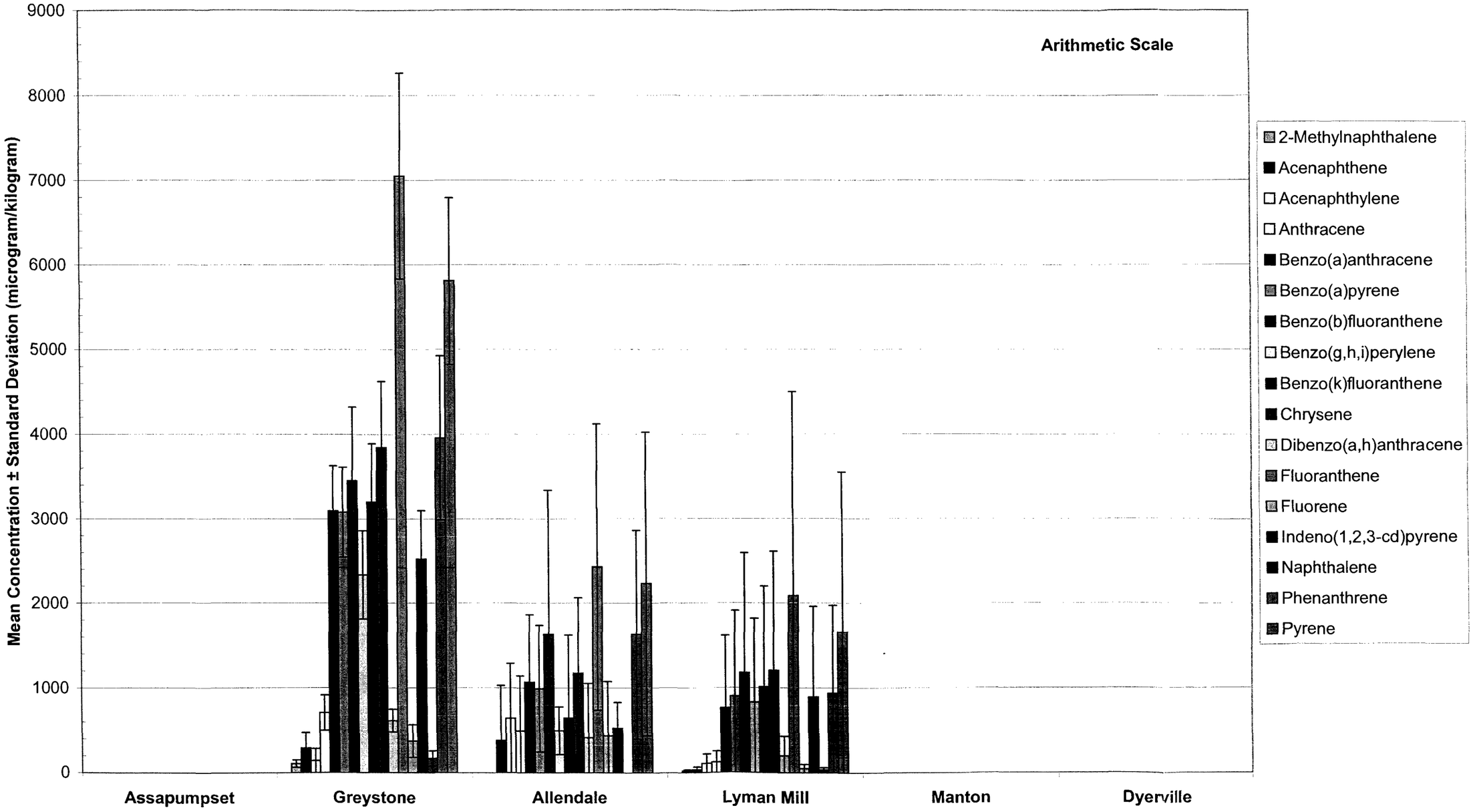
MACTEC Engineering and Consulting, Inc.
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Figure 30
Select PAH Concentrations in Floodplain and Bank Soil

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



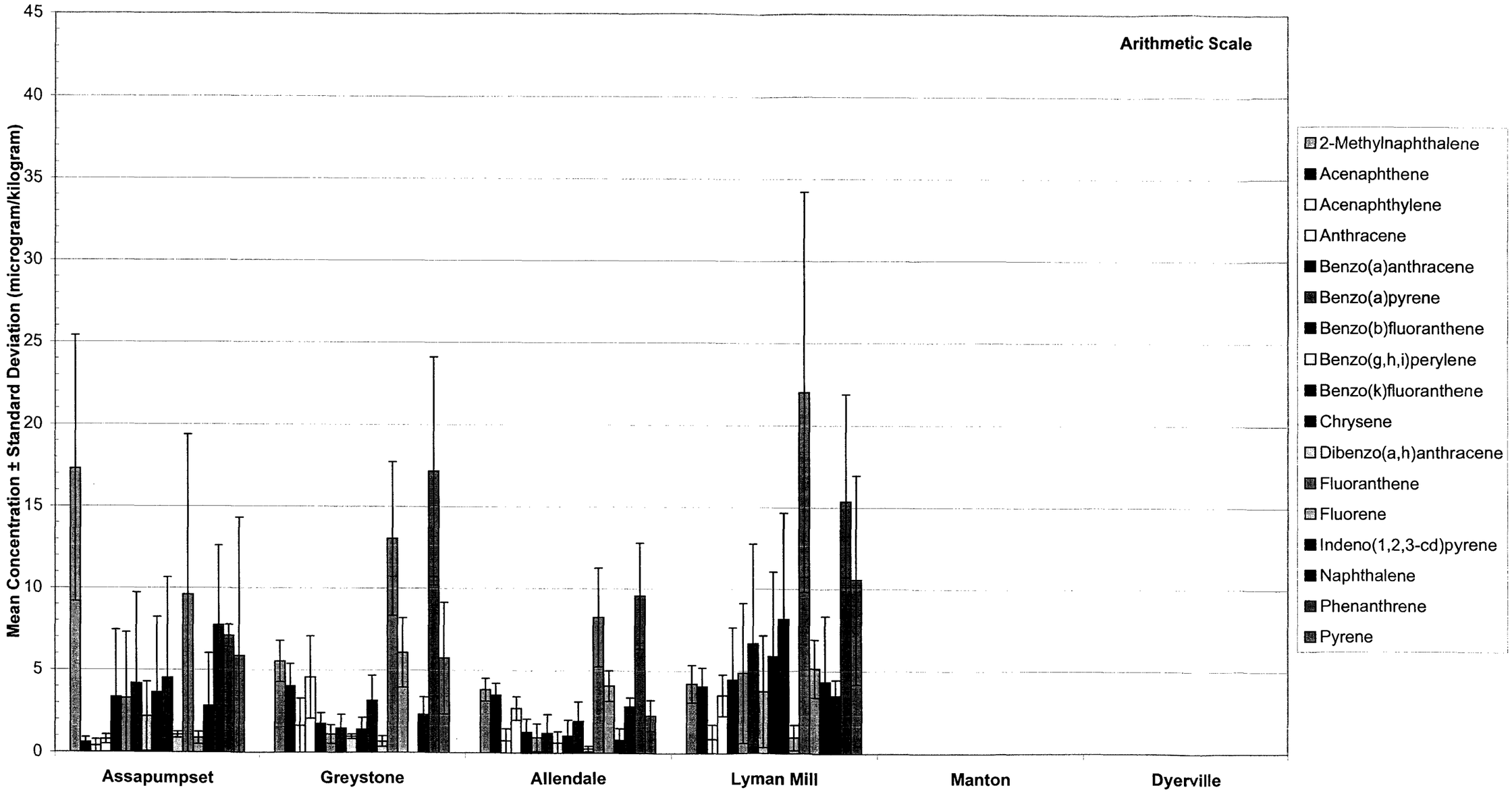
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Figure 31
Select PAH Concentrations in White Sucker

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Note: Brown Bullhead were used as a surrogate for White Sucker in Assapumpset Brook

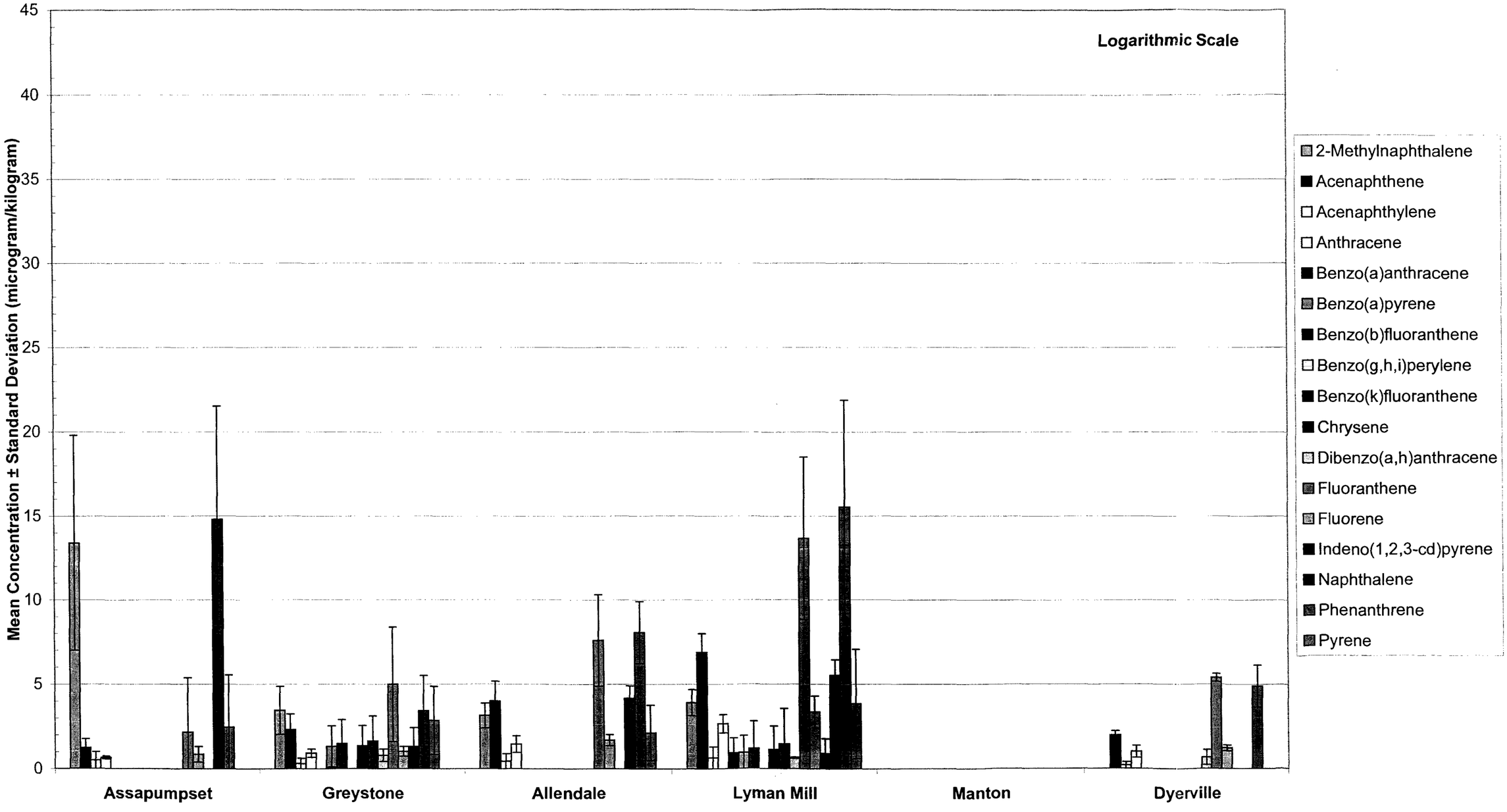
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Figure 32
Select PAH Concentrations in American Eel

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



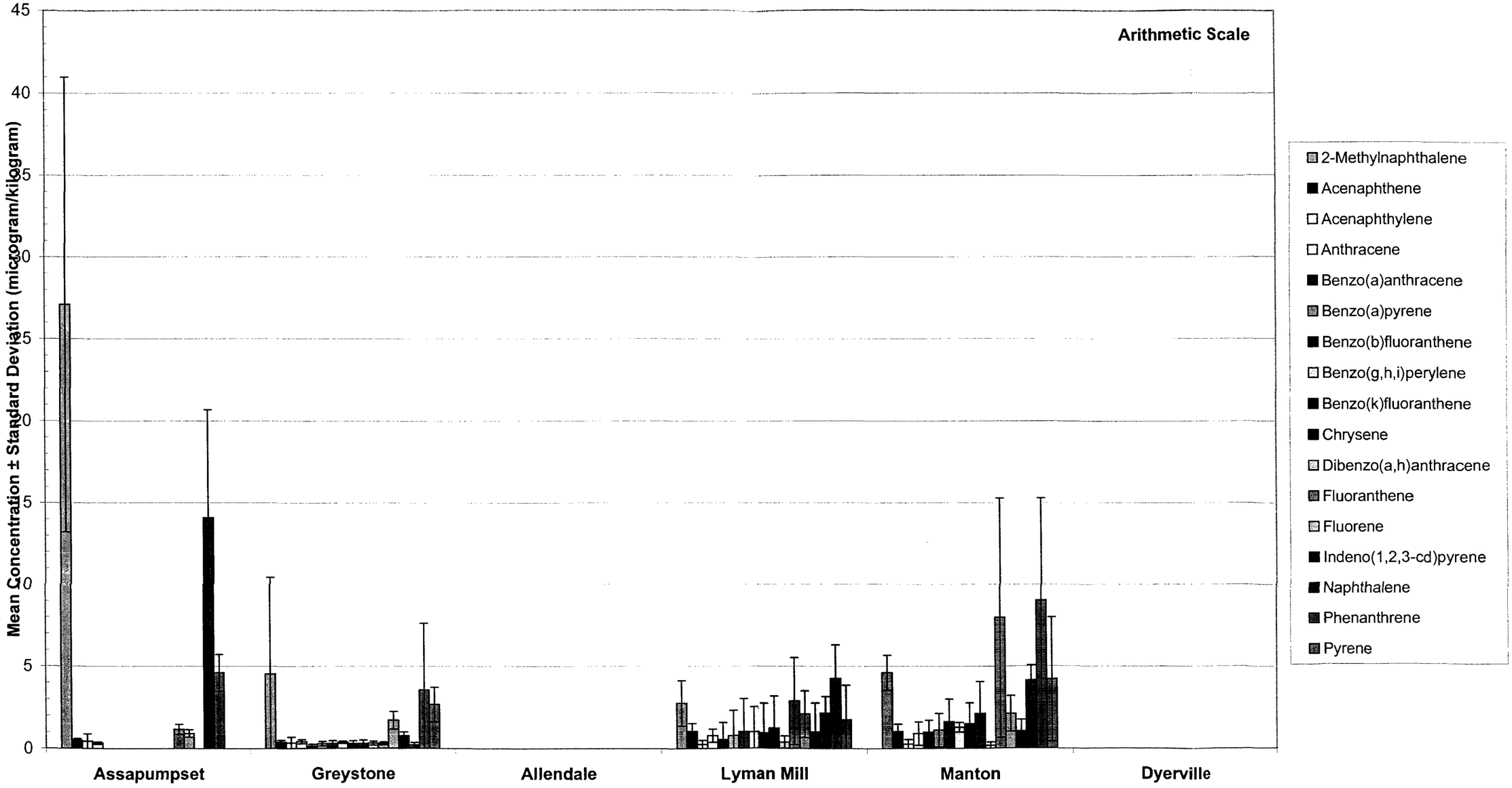
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Figure 33
Select PAH Concentrations in Largemouth Bass

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Reconstituted whole body data for Largemouth Bass used in developing histograms.

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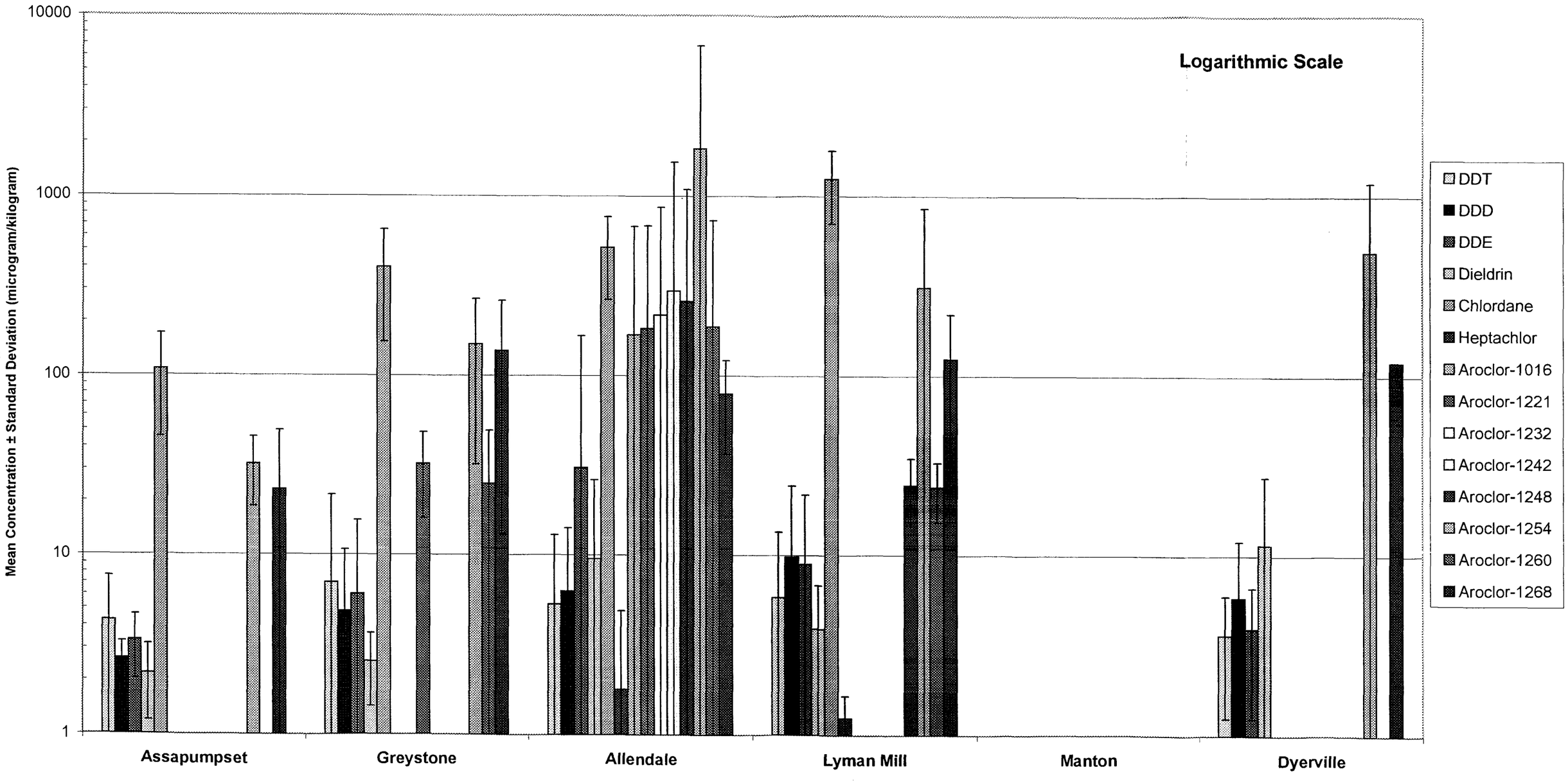
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Figure 34
Select Pesticide and PCB Aroclor Concentrations in Sediment

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

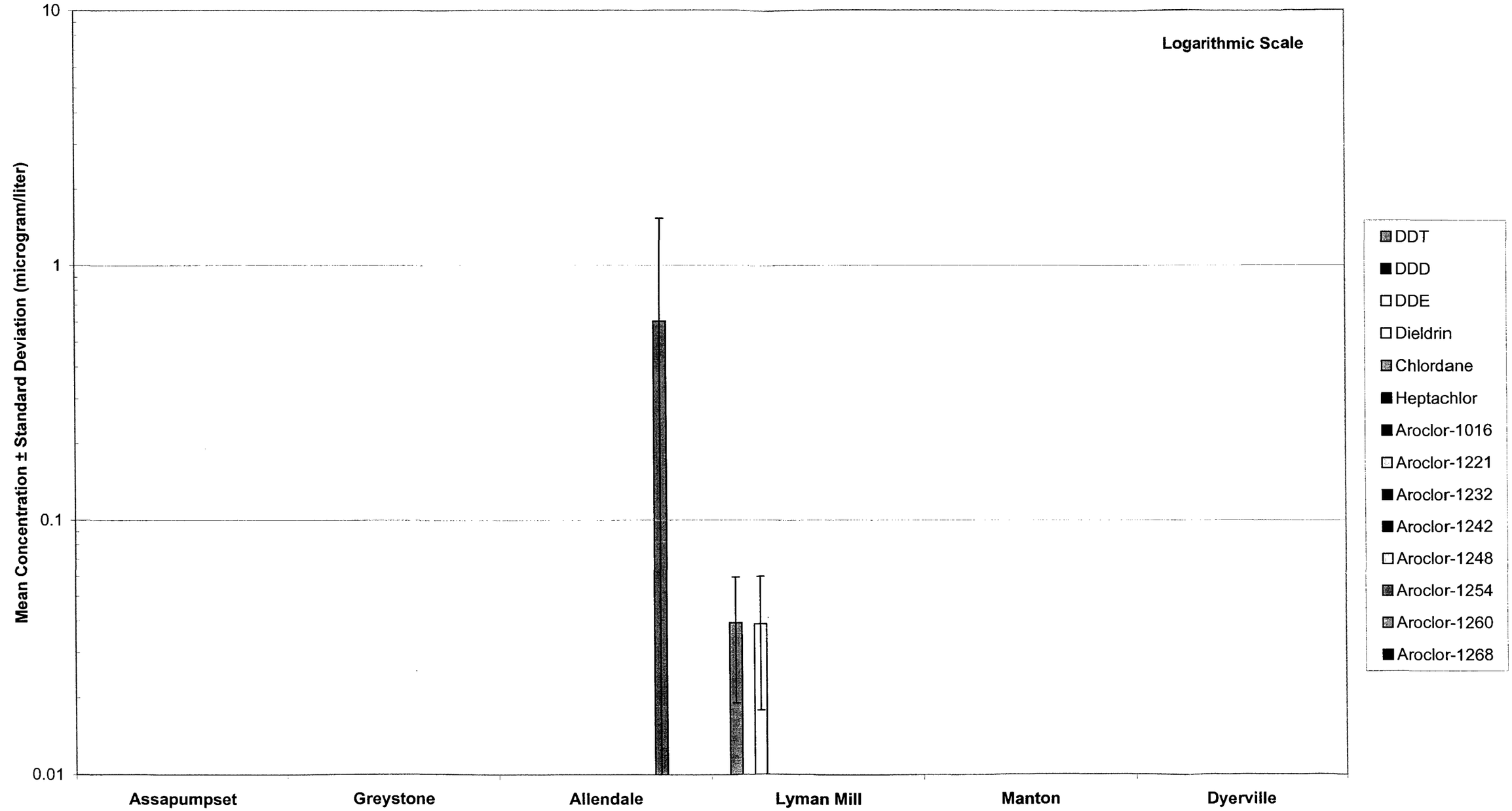


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Figure 35
Select Pesticide and PCB Aroclor Concentrations in Surface Water

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



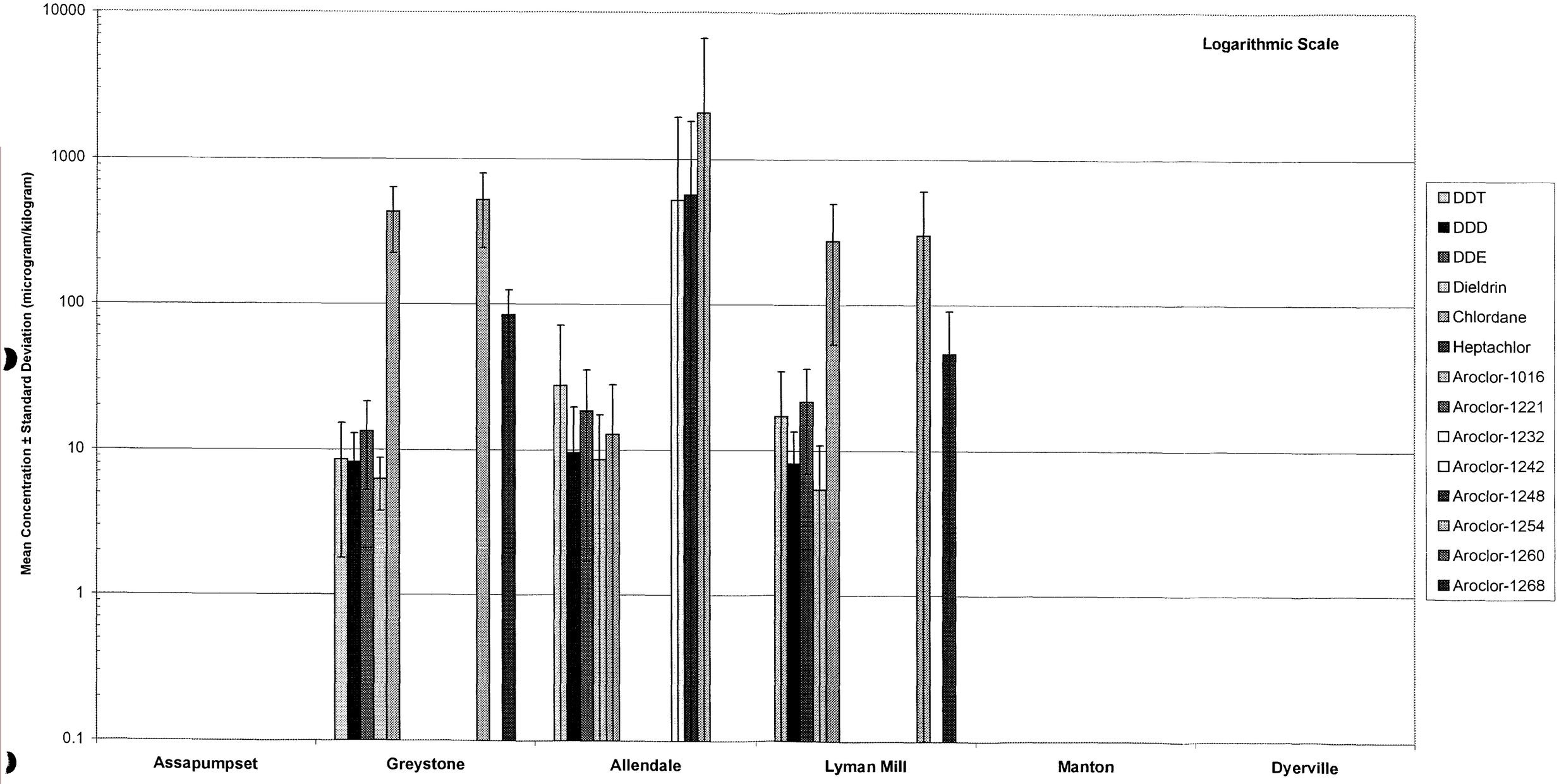
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Figure 36
Select Pesticide and PCB Aroclor Concentrations in Floodplain and Bank Soil

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



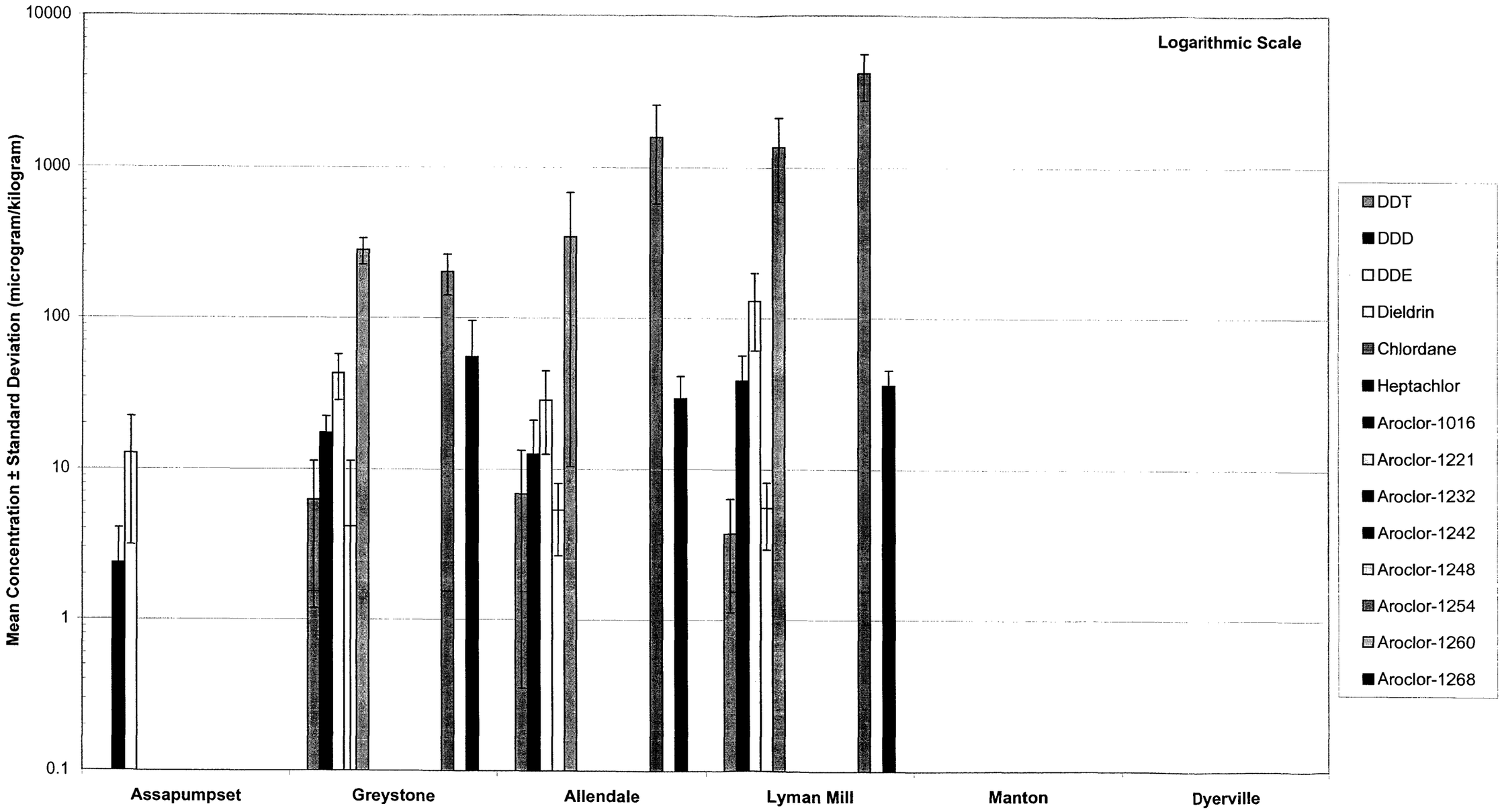
MACTEC Engineering and Consulting, Inc.
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Figure 37
Select Pesticide and PCB Aroclor Concentrations in White Sucker

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Note: *Brown Bullhead used as a surrogate for White Sucker in Assapumpset Brook*

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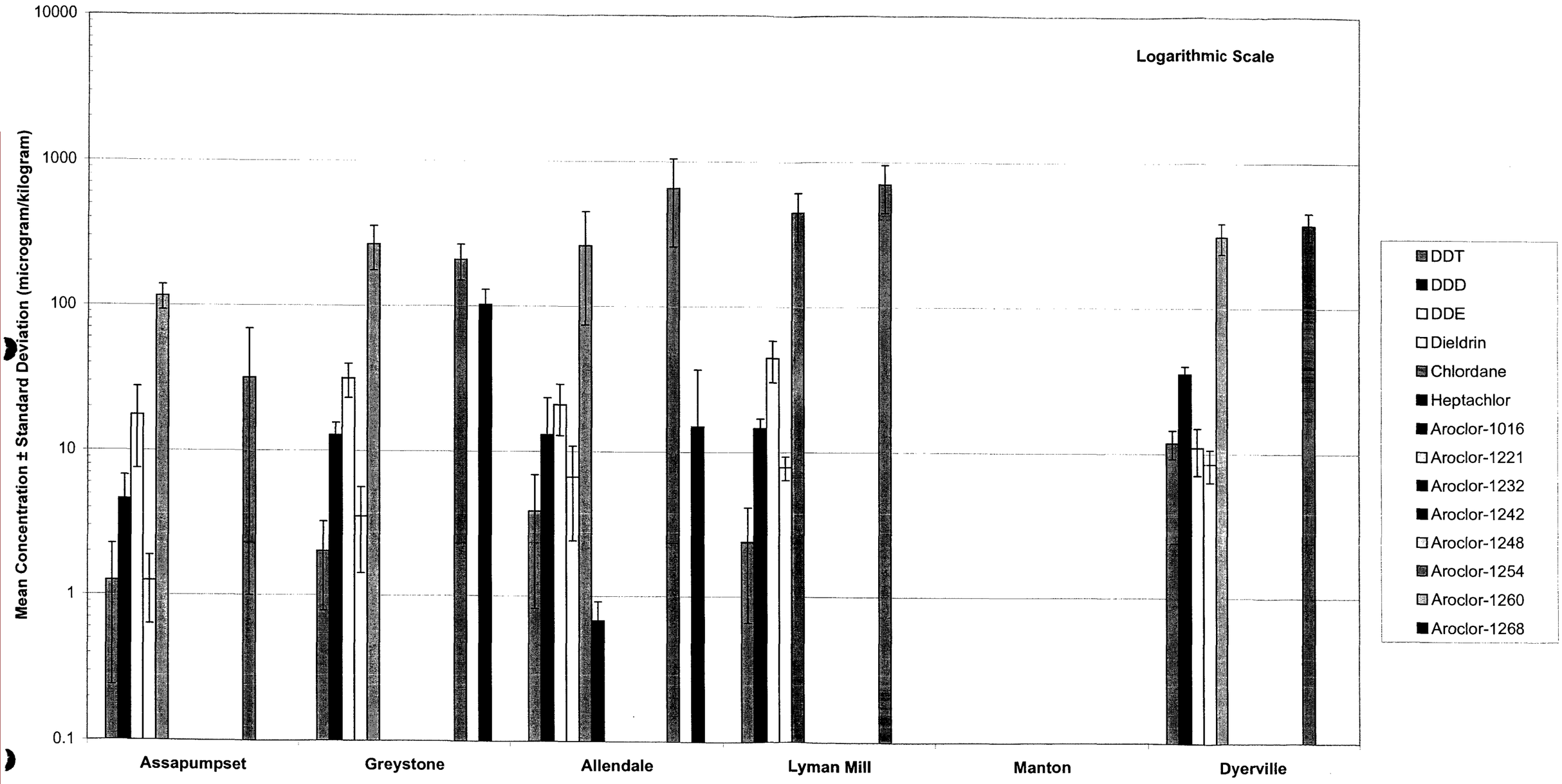
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Figure 38
Select Pesticide and PCB Aroclor Concentrations in American Eel

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

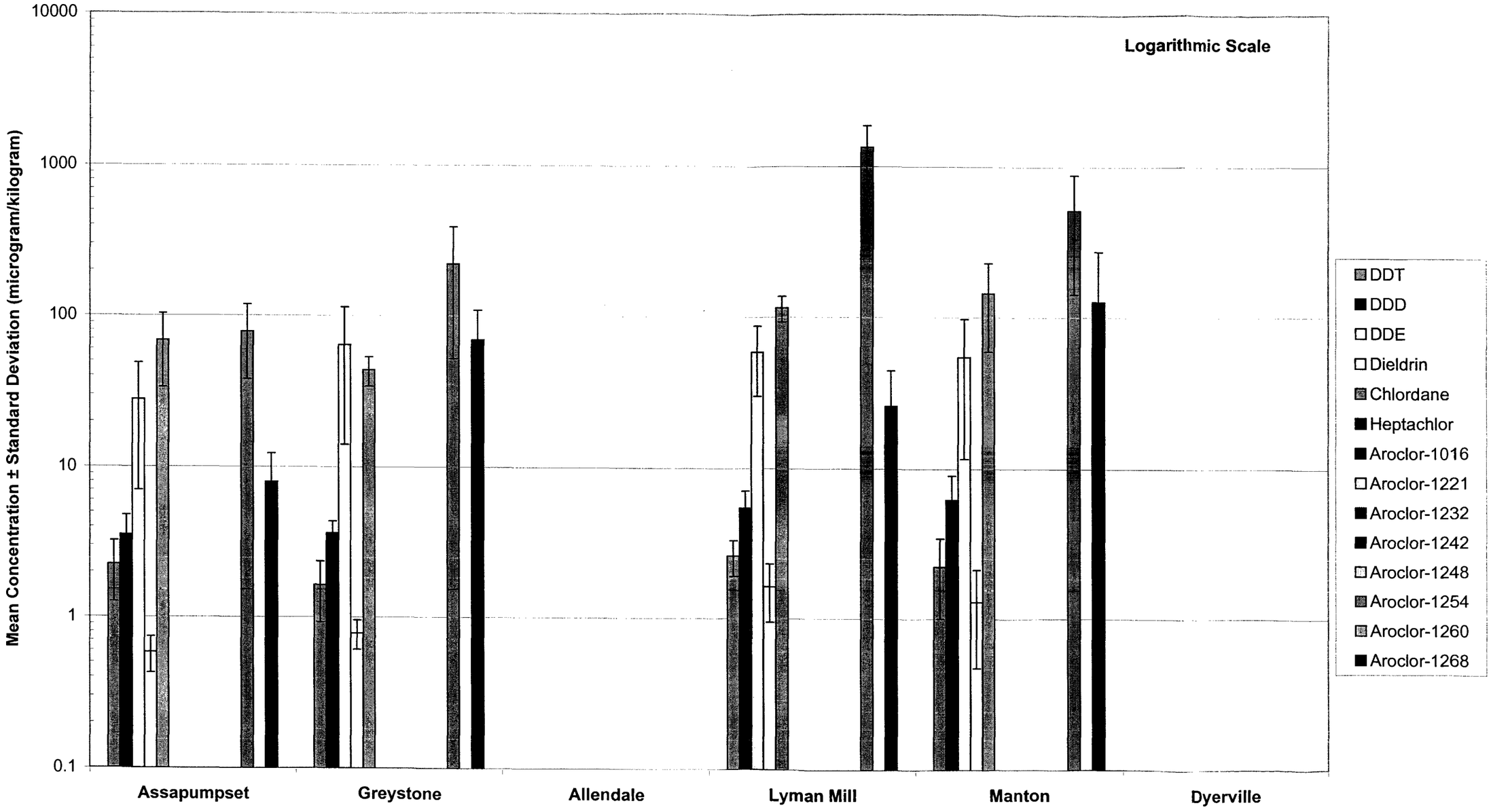


MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Eel, Pest-Aroclors Chart 1

8/5/2004

Figure 39
Select Pesticide and PCB Aroclor Concentrations in Largemouth Bass

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Reconstituted whole body data for Largemouth Bass used in developing histograms.

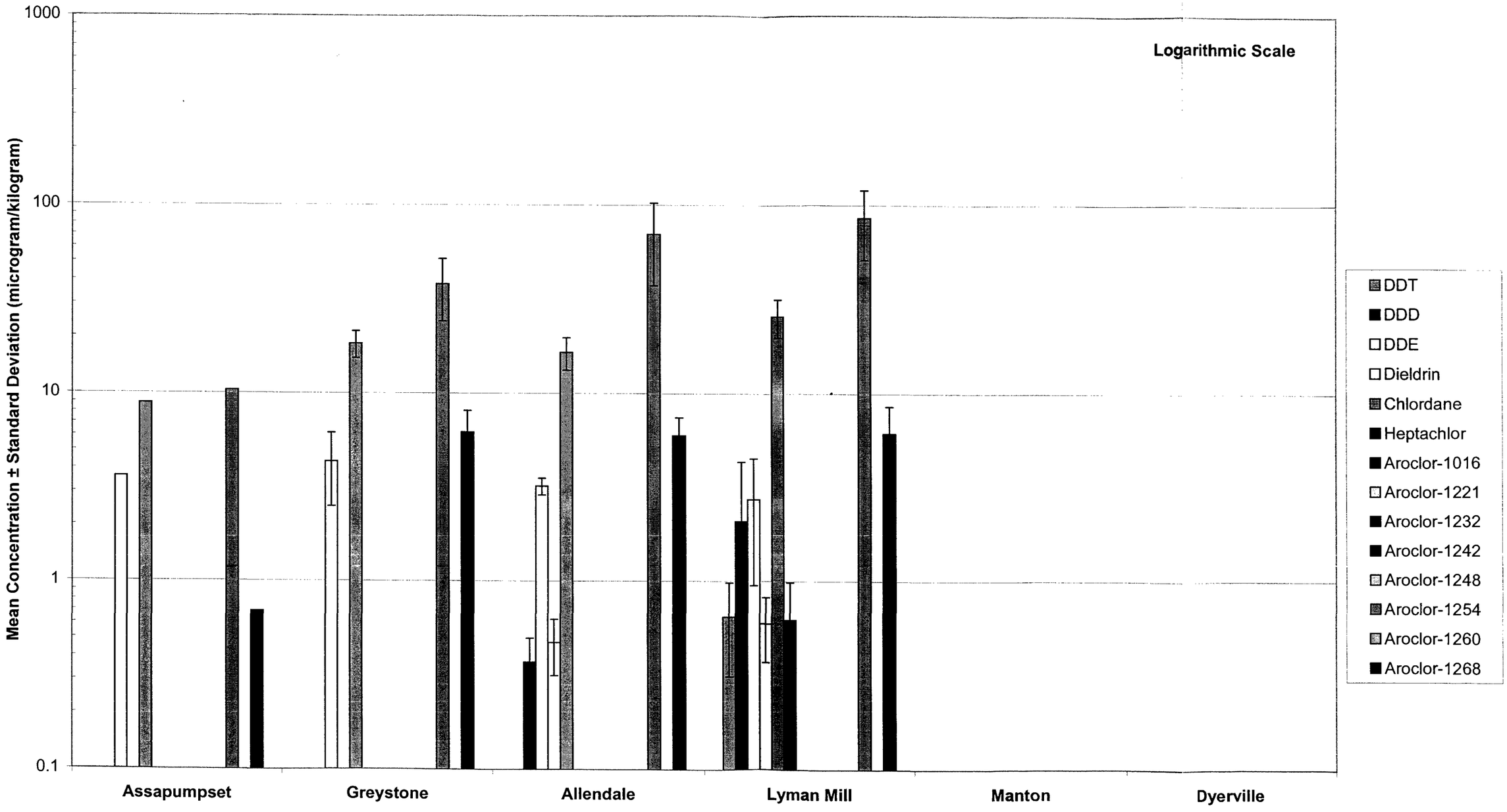
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Bass, Pest-Aroclors Chart 1

8/5/2004

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Figure 40
Select Pesticide and PCB Aroclor Concentrations in Crayfish

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



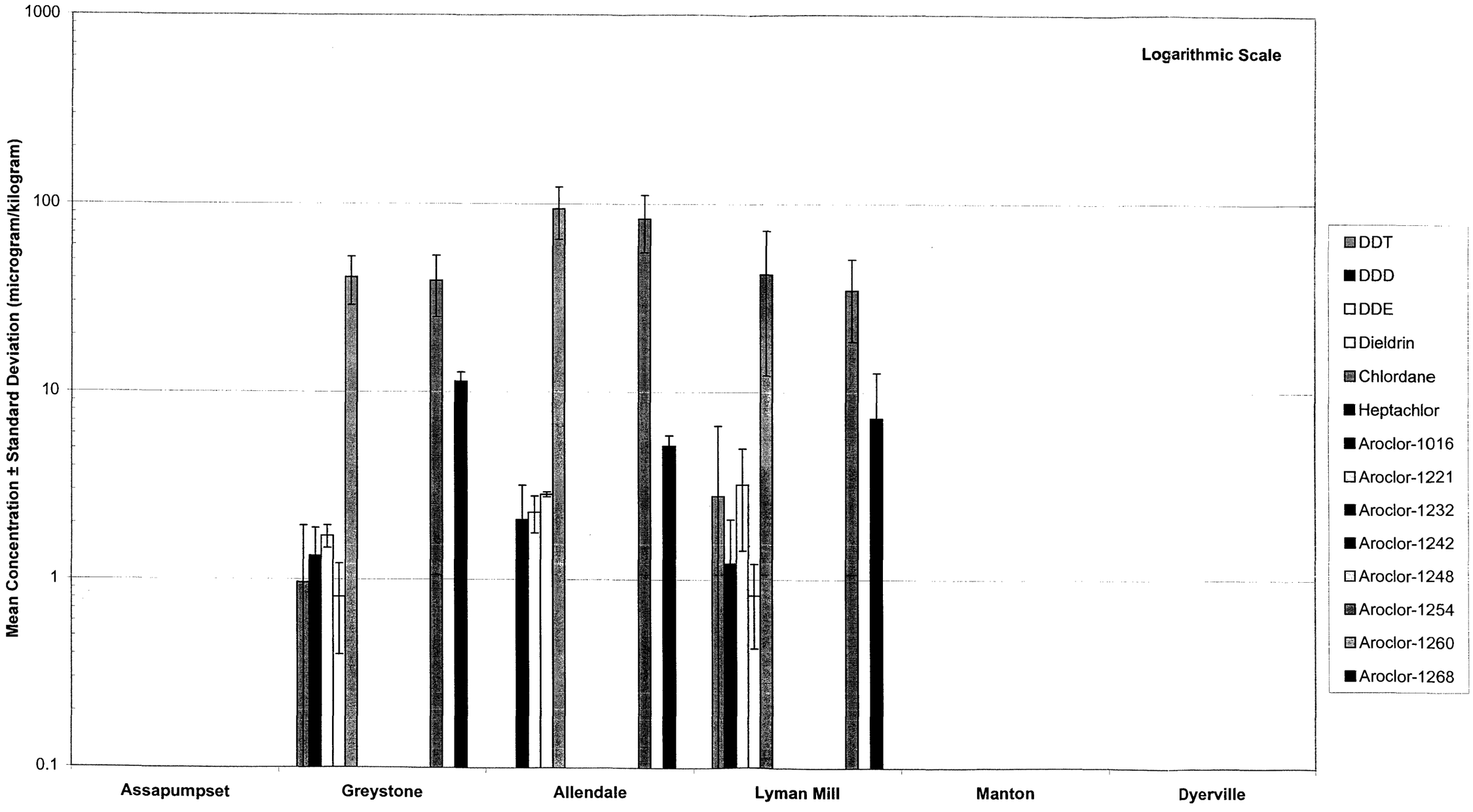
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Crayfish, Pest-Aroclors Chart 1

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Figure 41
Select Pesticide and PCB Aroclor Concentrations in Earthworms

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



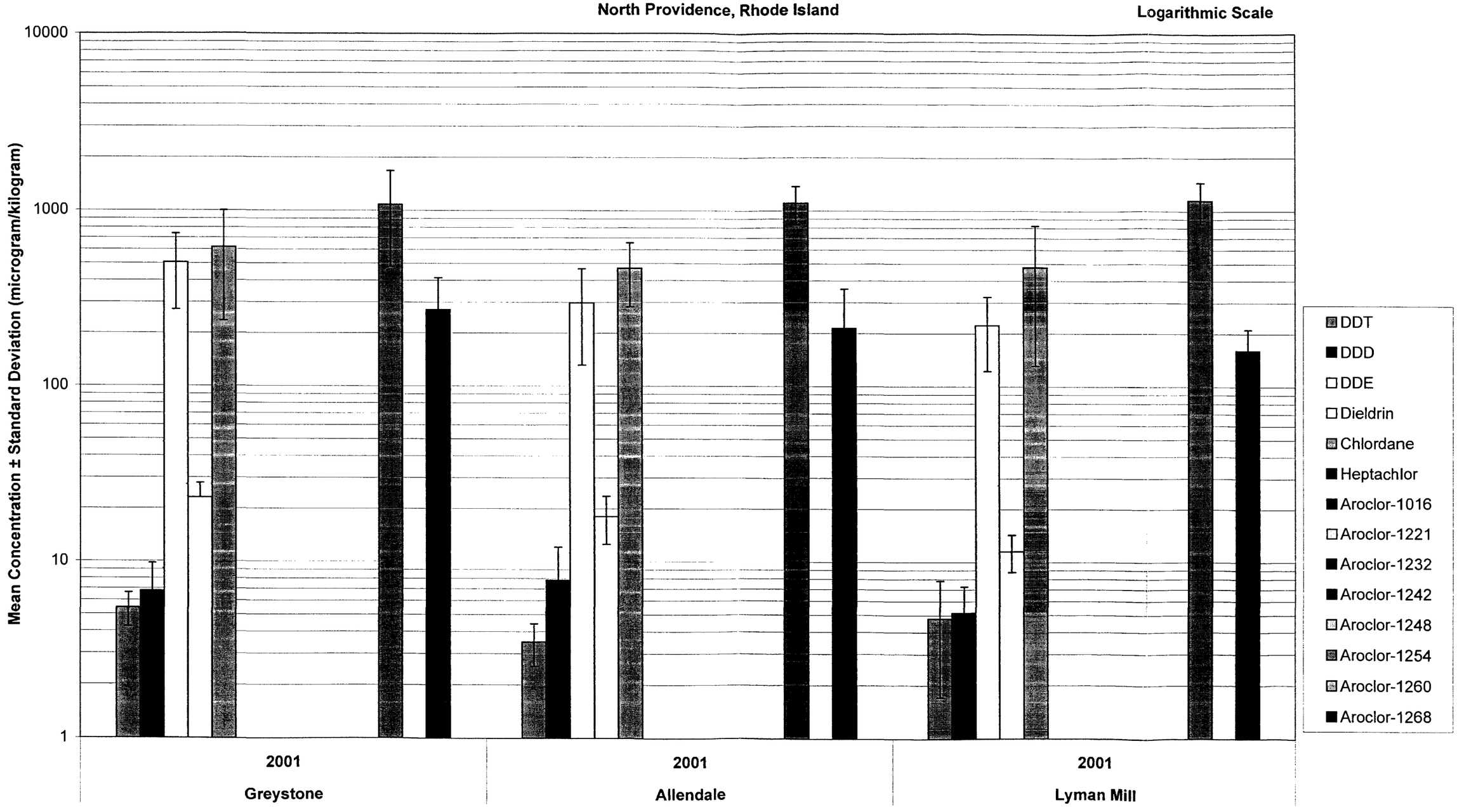
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Earthworm, Pest-Aroclors Chart 1

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Figure 42
Select Pesticide and PCB Aroclor Concentrations in Tree Swallow Egg

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



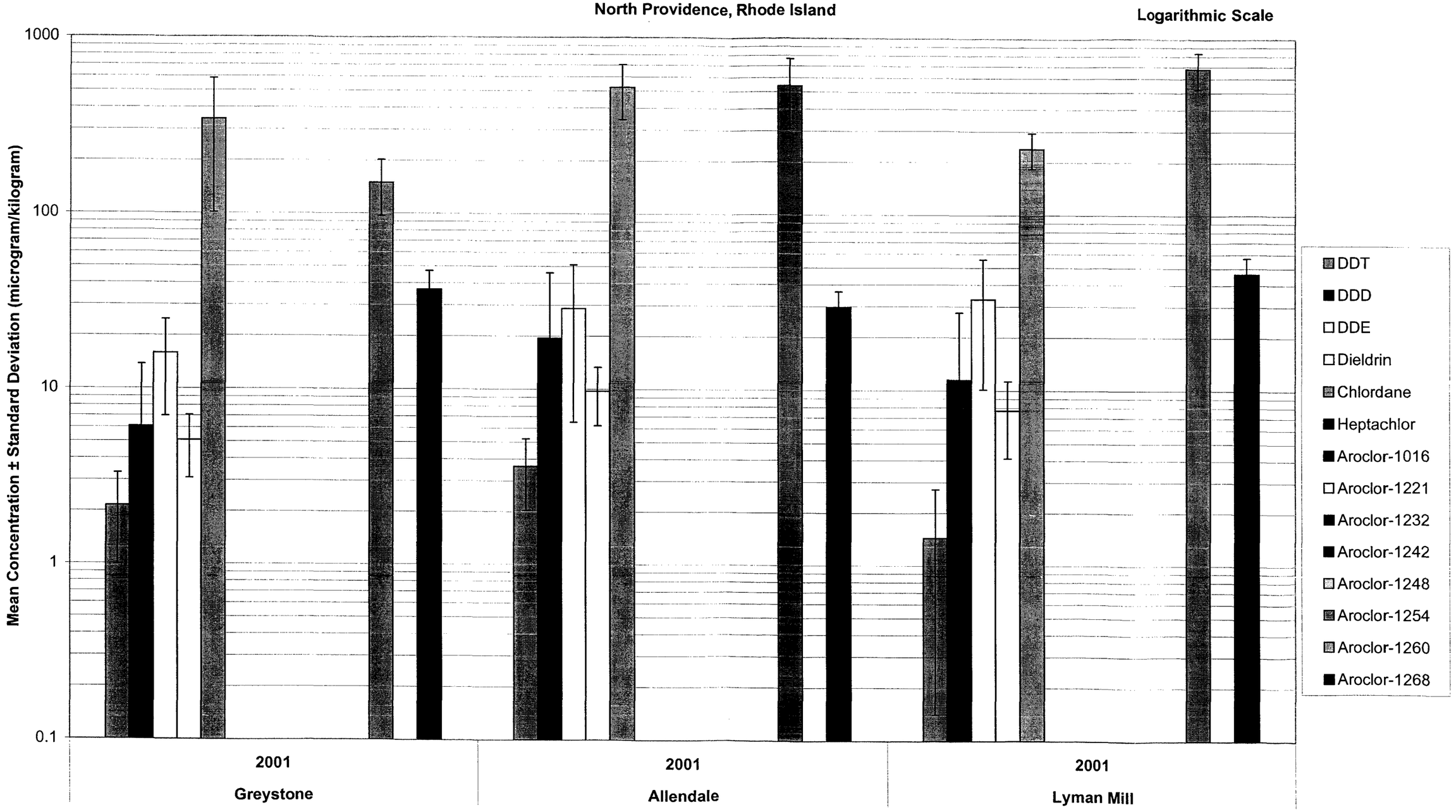
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\TreeSwallowEgg(NoSwitch).xls, Pest-Aroclors Chart 1

9/10/2004

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Figure 43
Select Pesticide and PCB Aroclor Concentrations in Tree Swallow Nestling

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Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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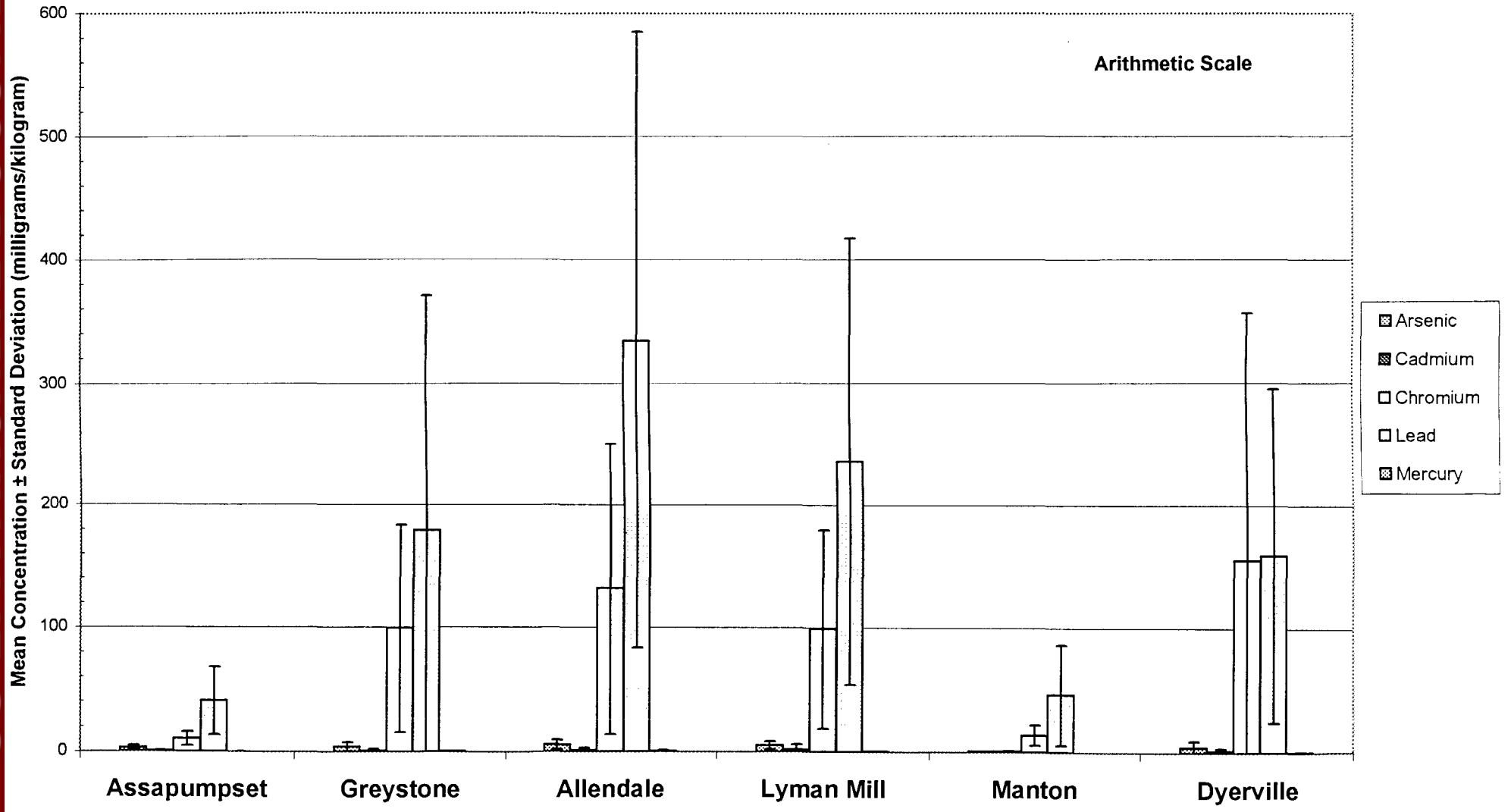
8/5/2004

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Figure 44
Select Inorganic Analyte Concentrations in Sediment

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



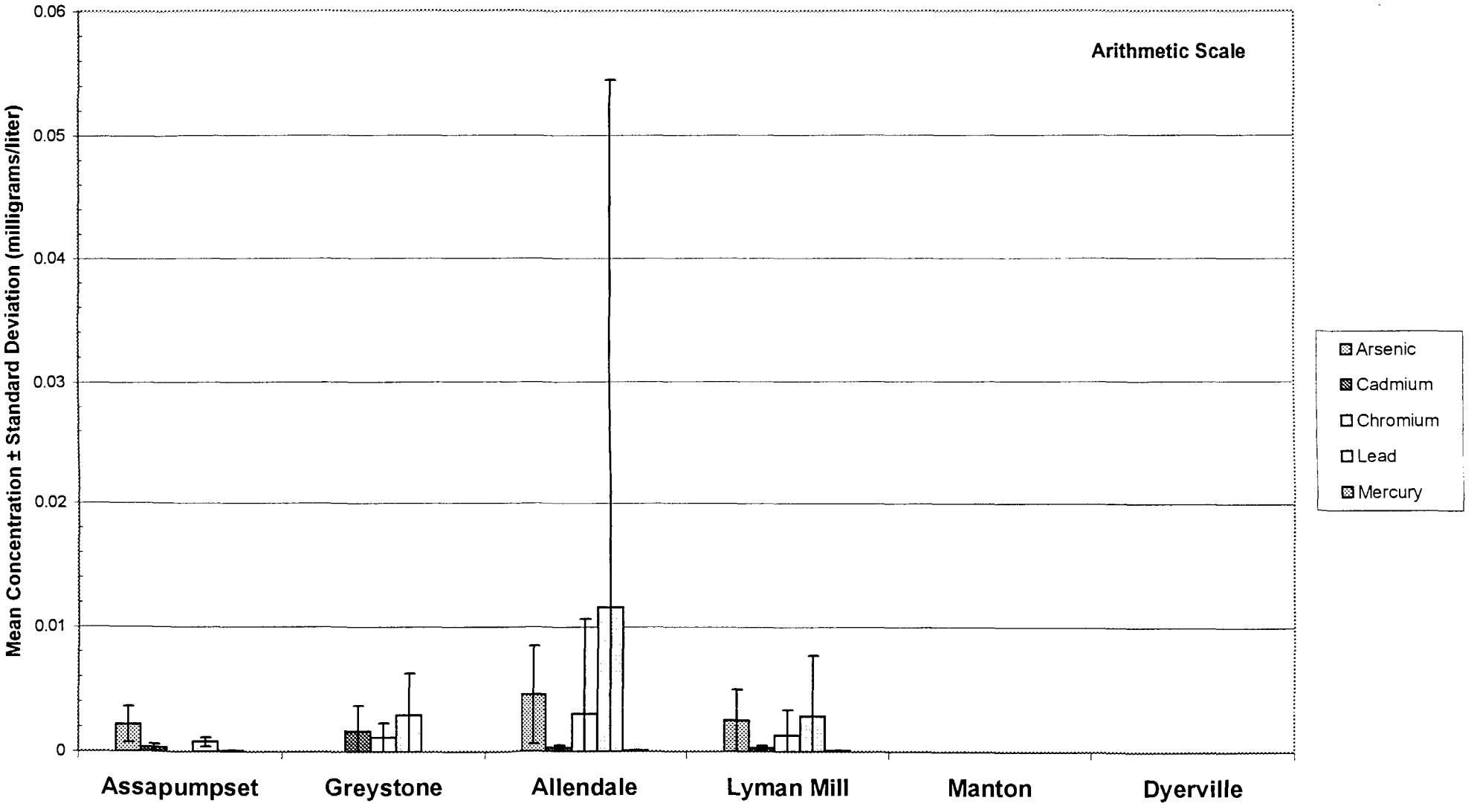
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\SED, Metals Chart 1

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Figure 45
Select Inorganic Analyte Concentrations in Surface Water

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

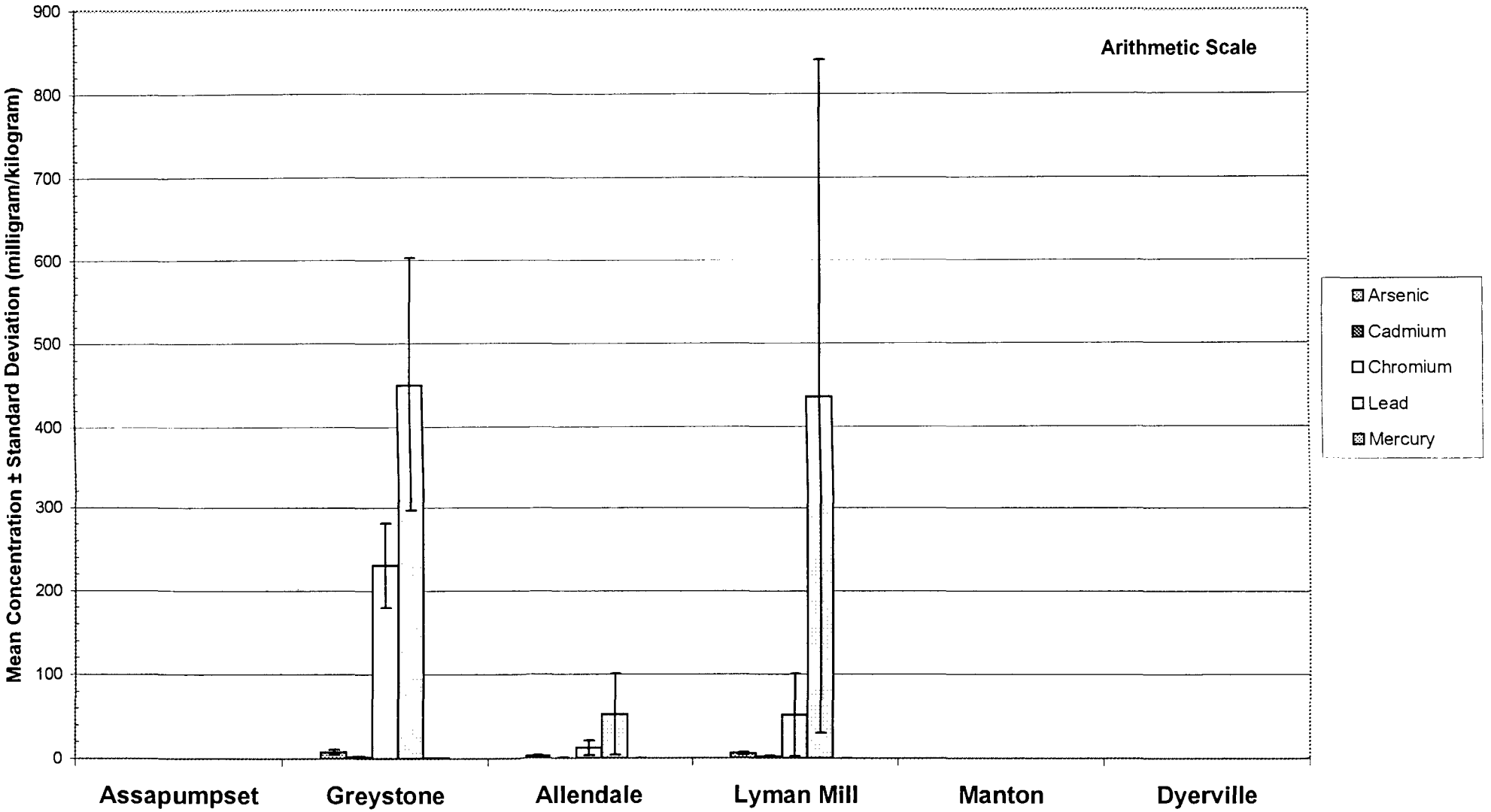


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51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\SW, Metals Chart 1

8/5/2004

Figure 46
Select Inorganic Analyte Concentrations in Floodplain and Bank Soil

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



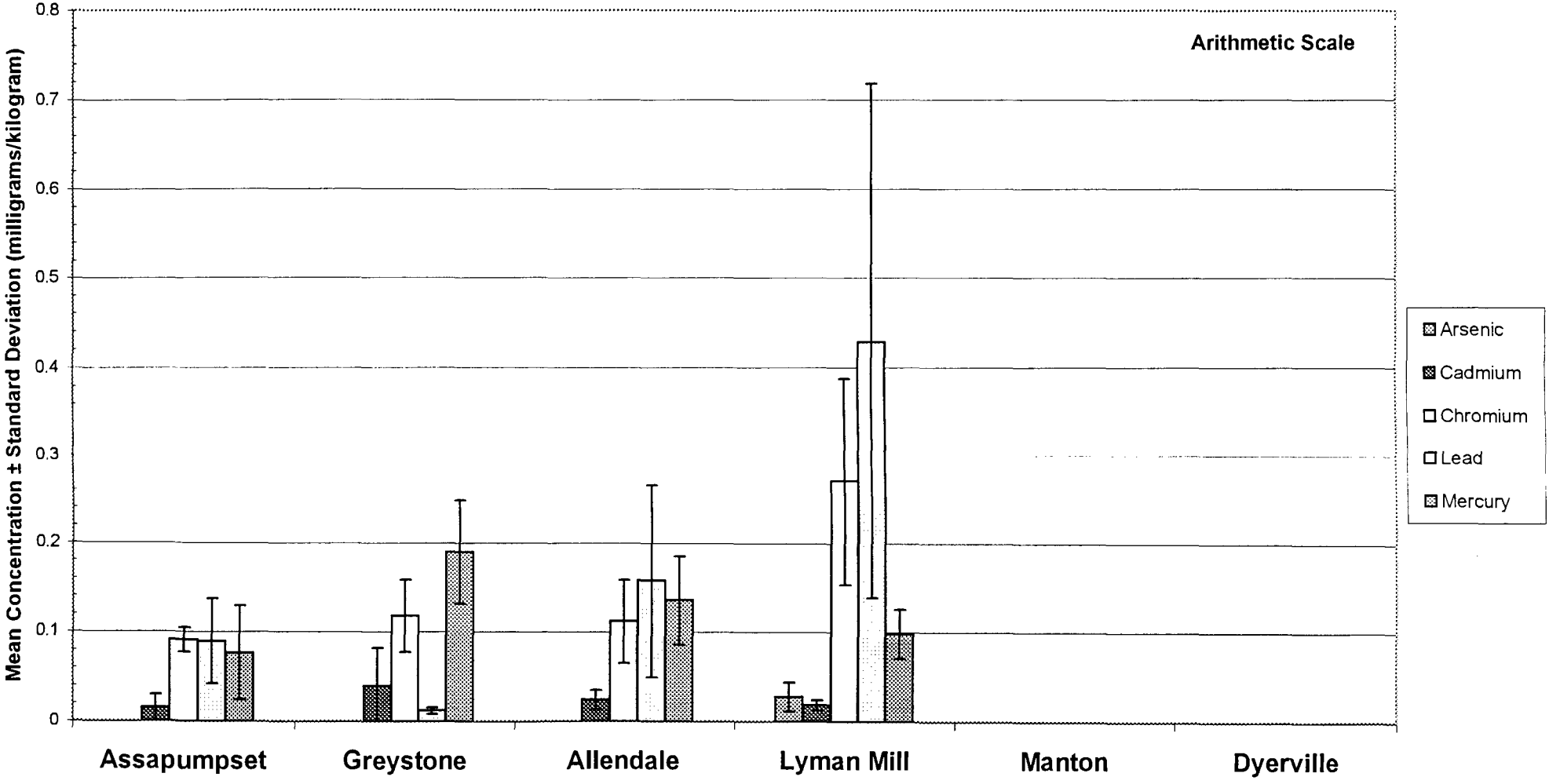
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\FP-Bank-Soil, Metals Chart 1

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Figure 47
Select Inorganic Analyte Concentrations in White Sucker

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

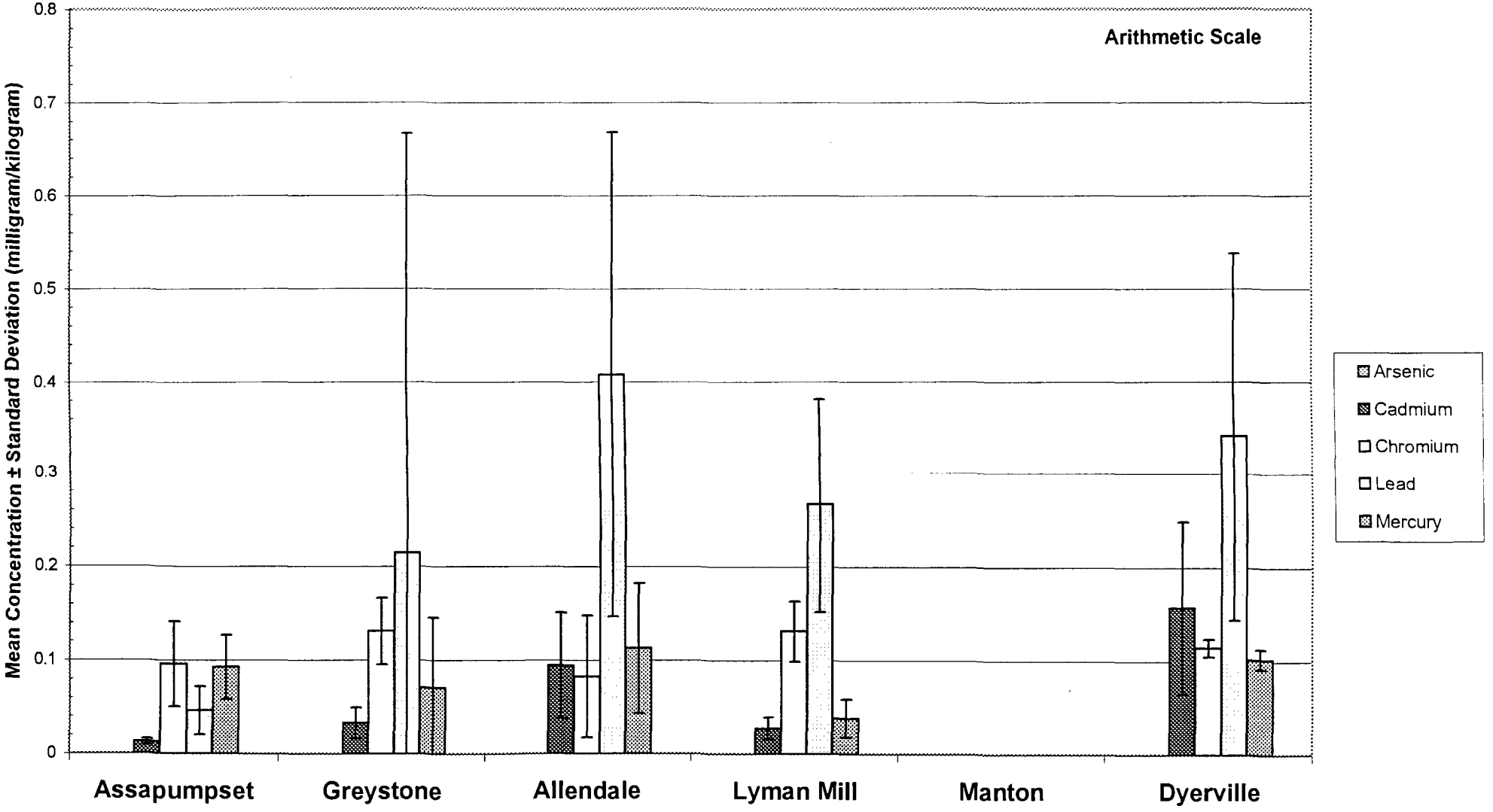


Note: Brown Bullhead used as a surrogate for White Sucker in Assapumpset Brook

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Figure 48
Select Inorganic Analyte Concentrations in American Eel

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



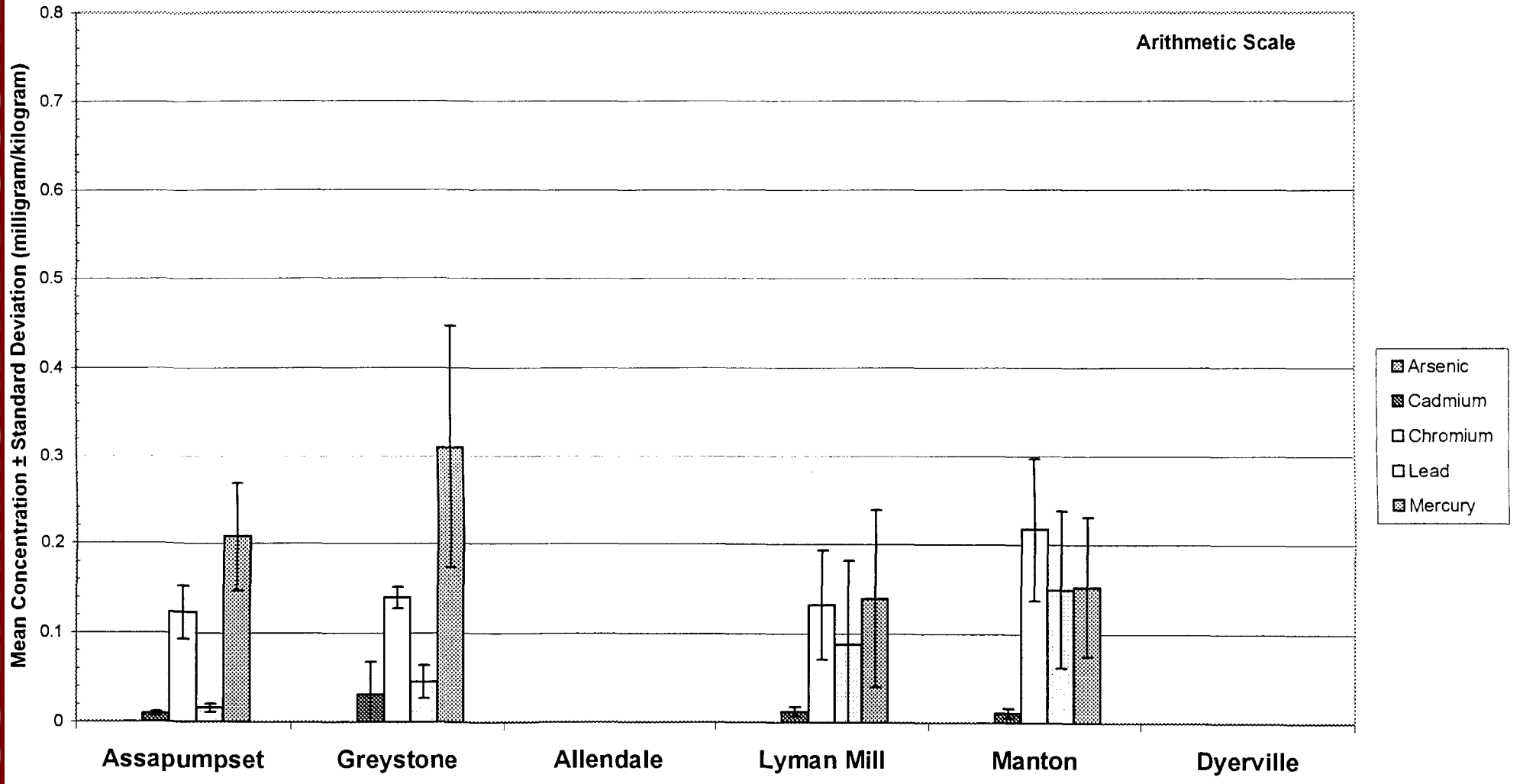
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51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Eel, Metals Chart 1

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Figure 49
Select Inorganic Analyte Concentrations in Largemouth Bass

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



Reconstituted whole body data for Largemouth Bass used in developing histograms.

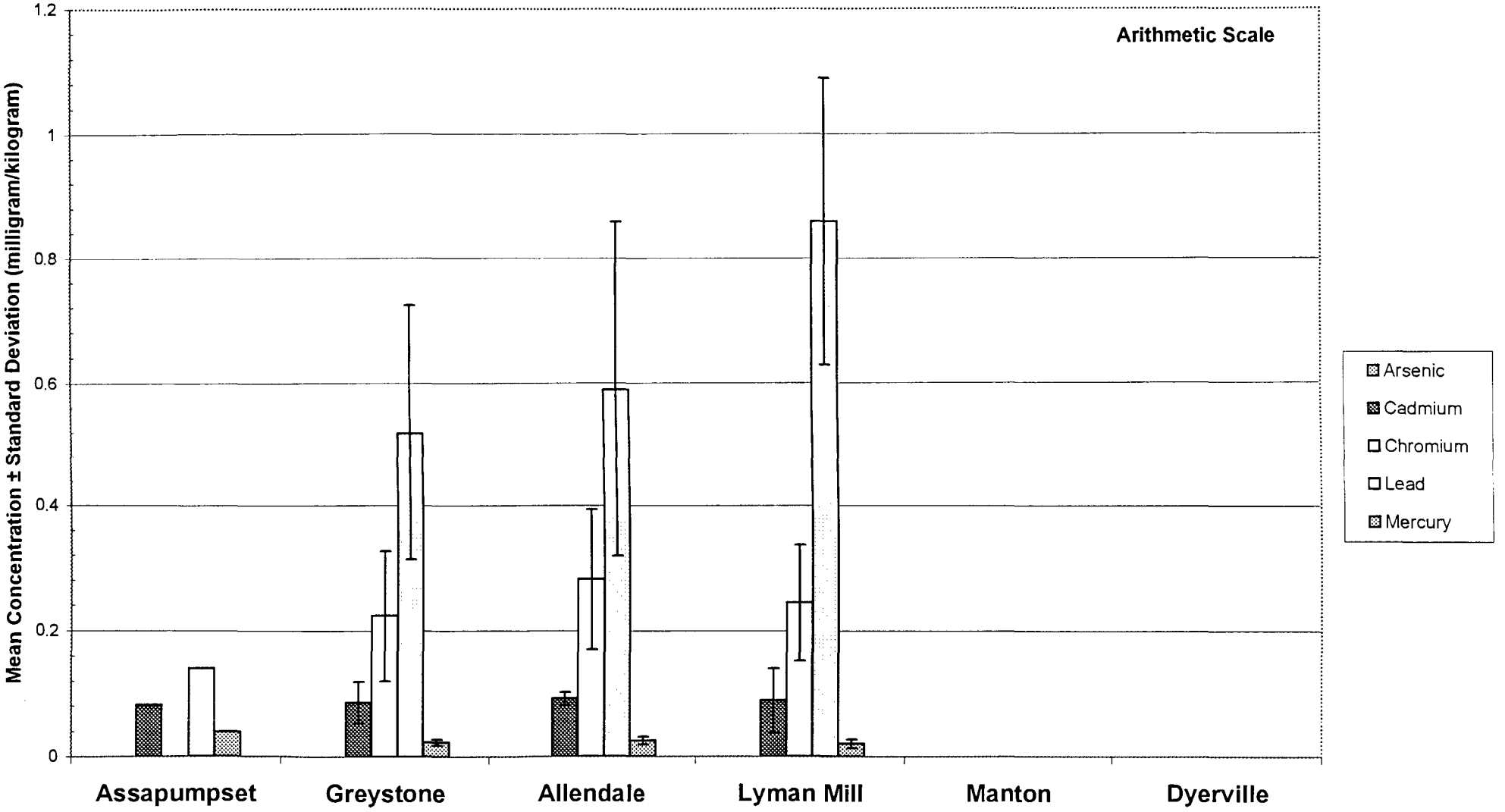
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51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Bass, Metals Chart 1

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Figure 50
Select Inorganics Concentrations in Crayfish

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



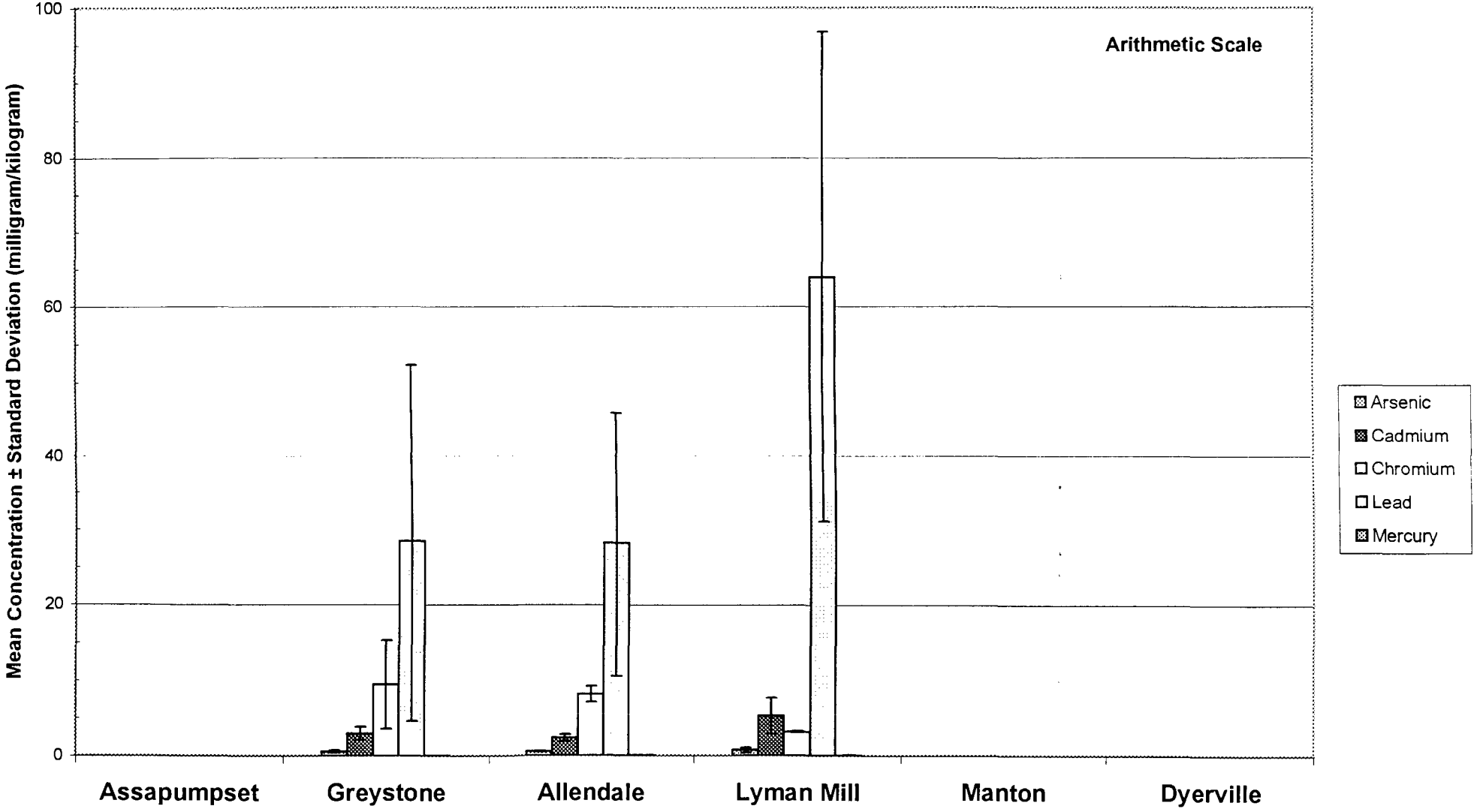
MACTEC Engineeering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\Histograms\Crayfish, Metals Chart 1

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Figure 51
Select Inorganic Analyte Concentrations in Earthworms

Baseline Ecological Risk Assessment
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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8/5/2004



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Figure 52
Distribution of Dioxin TEQ Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-TE

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Figure 53
Distribution of HCX Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-HCX.SRF

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GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected
  Symbol Size Proportional to Concentration
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 11 to 27,000 ug/Kg

MACTEC

Figure 54
Distribution of Total PCB Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-PCB.SRF

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GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected (Relative Concentration)
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 49 to 113,110 ug/Kg

MACTEC

Figure 55
Distribution of Total PAH Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-PAH.SRF

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Figure 56
Distribution of Total Pesticide Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-PEST.S

US EPA ARCHIVE DOCUMENT

GREYSTONE MILL POND DAM

ALLENDALE POND

ALLENDALE DAM

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

MANTON POND

MANTON DAM

WOONASQUATUCKET RIVER

DYERVILLE POND

DYERVILLE DAM

LEGEND

- Constituent Detected
  Symbol Size Proportional to Concentration
- Constituent Not Detected
- Constituent Not Sampled

SCALE IN FEET:

0 300 600

MAGNETIC

Range of Detected Concentrations: 0.938 to 333 Metals Toxic Units

MACTEC

Figure 57
Distribution of Metals (Toxic Units) Concentrations in Sediment
Baseline Ecological Risk Assessment

Prepared By: AWS
Checked By: KJA
Approved By: MJM

P:/W9-GVT/COE-NAE/BATTELLE/CENTREDALE/TASK25-BCRA/FIGURES/SURFER/ECO SD-MTU.SR

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Legend:

- SD/CF = Paired Sediment and Crayfish Locations
- EI = Emerging Insect Traps
- FP/EW = Paired Floodplain Soil and Earthworm Locations
- SD = Sediment Bioassay Locations
- General area where SD/CF composite sample was collected

GMP-EI-5001
GMP-EI-5002
GMP-EI-5005
GMP-SD-5002
GMP-CF-5002
GMP-EI-5003
GMP-EI-5004
GMP-SD-5007

GMP-SD-5007-0000-01 7/9/2001 Effects	
H.azteca 42-d survival	98%
H.azteca 42-d growth	0.325
H.azteca 42-d average # young/female	7
HQs	
Acenaphthene	9.4E00J
Anthracene	1.6E01J
Dibenzofuran	3.4E01J
Fluorene	1.5E01J
Phenanthrene	1.2E01J
Dieldrin	2.3E02J
Mercury	3.9E00J
PAH ESGTot (11.5-fold UF)	3.57
C-Normalized Excess SEM	14.09

GMP-SD-5007-0000-01 7/9/2001 Effects	
C. tentans 49-d survival	35%*
C. tentans 20-d growth	2.173
HQs	
Dieldrin	2.3E+02J
Aroclor-1254	2.7E+00
Mercury	3.9E+00J
PAH ESGTot (11.5-fold UF)	3.57
C-Normalized Excess SEM	14.09

RWR-SD-5003
RWR-CF-5003
RWR-FP-5003
RWR-EW-5003
RWR-FP-5004
RWR-EW-5004
RWR-SD-5004
RWR-CF-5004
RWR-FP-5002
RWR-EW-5002
RWR-FP-5001
RWR-EW-5001

APB-SD-4010-0000-01 7/10/2001 Effects	
C. tentans 49-d survival	0%*
C. tentans 20-d growth	-*
HQs	
Dieldrin	5.6E+03J
Aroclor-1254	5.4E+01
Mercury	6.2E+00J
PAH ESGTot (11.5-fold UF)	29.9
C-Normalized Excess SEM	-41.8

APB-SD-4010-0000-01 7/10/2001 Effects	
H.azteca 42-d survival	74%*
H.azteca 42-d growth	0.494
H.azteca 42-d average # young/female	6.8
HQs	
Acenaphthene	1.6E01
Anthracene	1.6E01
Dibenzofuran	4.1E01
Fluorene	1.9E01
Phenanthrene	1.6E01
Dieldrin	5.6E03J
Mercury	6.2E00J
PAH ESGTot (11.5-fold UF)	29.9
C-Normalized Excess SEM	-41.8

APB-SD-4010
CMS-FP-4001
CMS-EW-4001
CMS-FP-4005
CMS-EW-4005
CMS-FP-4002
CMS-EW-4002
CMS-SD-4003
CMS-CF-4003
CMS-SD-4006 (COMPOSITE)
CMS-CF-4006 (COMPOSITE)
CMS-FP-4003
CMS-EW-4003
CMS-SD-4007 (COMPOSITE)
CMS-CF-4007 (COMPOSITE)

APB-SD-4008-0000-01 7/9/2001 Effects	
C. tentans 49-d survival	48%*
C. tentans 20-d growth	0.685*
HQs	
Dieldrin	2.5E02J
Aroclor-1254	1.1E+01
Mercury	4.5E+00J
PAH ESGTot (11.5-fold UF)	5.41
C-Normalized Excess SEM	144

APB-SD-4008-0000-01 7/9/2001 Effects	
H.azteca 42-d survival	83%*
H.azteca 42-d growth	0.315
H.azteca 42-d average # young/female	2.6*
HQs	
Acenaphthene	7.1E+00J
Anthracene	1.4E+01J
Dibenzofuran	2.8E-01J
Fluorene	1.1E+01J
Phenanthrene	1.2E+01J
Dieldrin	2.5E+02J
Mercury	4.5E+00J
PAH ESGTot (11.5-fold UF)	5.41
C-Normalized Excess SEM	144

APB-SD-4008
APB-SD-4009
WRL-SD-4005
WRL-CF-4005

APB-SD-4009-0000-01 7/9/2001 Effects	
H.azteca 42-d survival	80%*
H.azteca 42-d growth	0.418
H.azteca 42-d average # young/female	6.9
HQs	
Acenaphthene	1.8E+01
Anthracene	2.9E+01
Dibenzofuran	6.0E-01
Fluorene	2.3E+01
Phenanthrene	2.5E+01
Dieldrin	4.1E02J
Mercury	4.2E00J
PAH ESGTot (11.5-fold UF)	11.16
C-Normalized Excess SEM	10.5

APB-SD-4009-0000-01 7/9/2001 Effects	
C. tentans 49-d survival	32%*
C. tentans 20-d growth	1.9
HQs	
Dieldrin	4.1E+02J
Aroclor-1254	2.4E+01
Mercury	4.2E+00J
PAH ESGTot (11.5-fold UF)	11.16
C-Normalized Excess SEM	10.5

LPX-SD-4012-0000-01 7/10/2001 Effects	
C. tentans 49-d survival	50%*
C. tentans 20-d growth	1.729
HQs	
Dieldrin	3.1E+02J
Aroclor-1254	4.0E+00
Mercury	4.9E+00J
PAH ESGTot (11.5-fold UF)	7.71
C-Normalized Excess SEM	16.96

LPX-SD-4012-0000-01 7/10/2001 Effects	
H.azteca 42-d survival	78%*
H.azteca 42-d growth	0.438
H.azteca 42-d average # young/female	5.7
HQs	
Acenaphthene	2.0E01
Anthracene	4.2E01
Dibenzofuran	6.5E01
Fluorene	3.5E01
Phenanthrene	4.1E01
Dieldrin	3.1E02J
Mercury	4.9E00J
PAH ESGTot (11.5-fold UF)	7.71
C-Normalized Excess SEM	16.96

ASSAPUMPSET BROOK

LPX-FP-4004
LPX-EW-4004
RAB-SD-5004 (COMPOSITE)
RAB-CF-5004 (COMPOSITE)
LPX-SD-4012
LPX-SD-4001
LPX-CF-4001
WOONASQUATUCKET RIVER
LPX-EI-4006
LPX-EI-4007
LPX-FP-4006
LPX-EW-4006
LPX-SD-4002
LPX-CF-4002
LPX-EI-4008
LPX-SD-4013
LPX-FP-4007
LPX-EW-4007
LPX-EI-4009
LPX-EI-4010
LPX-SD-4004
LPX-CF-4004
LPX-SD-4011
RAB-SD-5008

RAB-SD-5008-0000-01 7/9/2001 Effects	
C. tentans 49-d survival	74%
C. tentans 20-d growth	1.78
HQs	
Dieldrin	9.6E+01J
Aroclor-1254	9.6E-01
Mercury	7.5E-01J
PAH ESGTot (11.5-fold UF)	3.34
C-Normalized Excess SEM	4.3

RAB-SD-5008-0000-01 7/9/2001 Effects	
H.azteca 42-d survival	94%
H.azteca 42-d growth	0.318
H.azteca 42-d average # young/female	9.3
HQs	
Acenaphthene	2.1E00J
Anthracene	5.5E00
Dibenzofuran	8.5E02
Fluorene	4.2E00
Phenanthrene	3.9E00
Dieldrin	9.6E01J
Mercury	7.5E01J
PAH ESGTot (11.5-fold UF)	3.34
C-Normalized Excess SEM	4.3

LPX-SD-4013-0000-01 7/10/2001 Effects	
C. tentans 49-d survival	52%
C. tentans 20-d growth	1.799
HQs	
Dieldrin	3.1E+02J
Aroclor-1254	2.1E+01
Mercury	1.9E+00J
PAH ESGTot (11.5-fold UF)	3.4
C-Normalized Excess SEM	58

LPX-SD-4013-0000-01 7/10/2001 Effects	
H.azteca 42-d survival	93%
H.azteca 42-d growth	0.331
H.azteca 42-d average # young/female	4.6*
HQs	
Acenaphthene	3.4E00J
Anthracene	8.4E00J
Dibenzofuran	1.8E01J
Fluorene	7.1E00J
Phenanthrene	6.0E00J
Dieldrin	3.1E02J
Mercury	1.9E00J
PAH ESGTot (11.5-fold UF)	3.45
C-Normalized Excess SEM	8

LPX-SD-4011-0000-01 7/10/2001 Effects	
C. tentans 49-d survival	48%*
C. tentans 20-d growth	1.937
HQs	
Dieldrin	3.4E+02J
Aroclor-1254	3.4E+01
Mercury	5.4E+00J
PAH ESGTot (11.5-fold UF)	4.95
C-Normalized Excess SEM	10.1

LPX-SD-4011-0000-01 7/10/2001 Effects	
H.azteca 42-d survival	91%
H.azteca 42-d growth	0.339
H.azteca 42-d average # young/female	5.6*
HQs	
Acenaphthene	7.2E00J
Anthracene	1.5E01J
Dibenzofuran	2.9E01J
Fluorene	1.3E01J
Phenanthrene	1.0E01J
Dieldrin	3.4E02J
Mercury	5.4E00J
PAH ESGTot (11.5-fold UF)	4.95
C-Normalized Excess SEM	10.1

400 0 400 800
SCALE IN FEET

MACTEC
107 Audubon Road
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Wakefield, MA 01880
781-245-6606

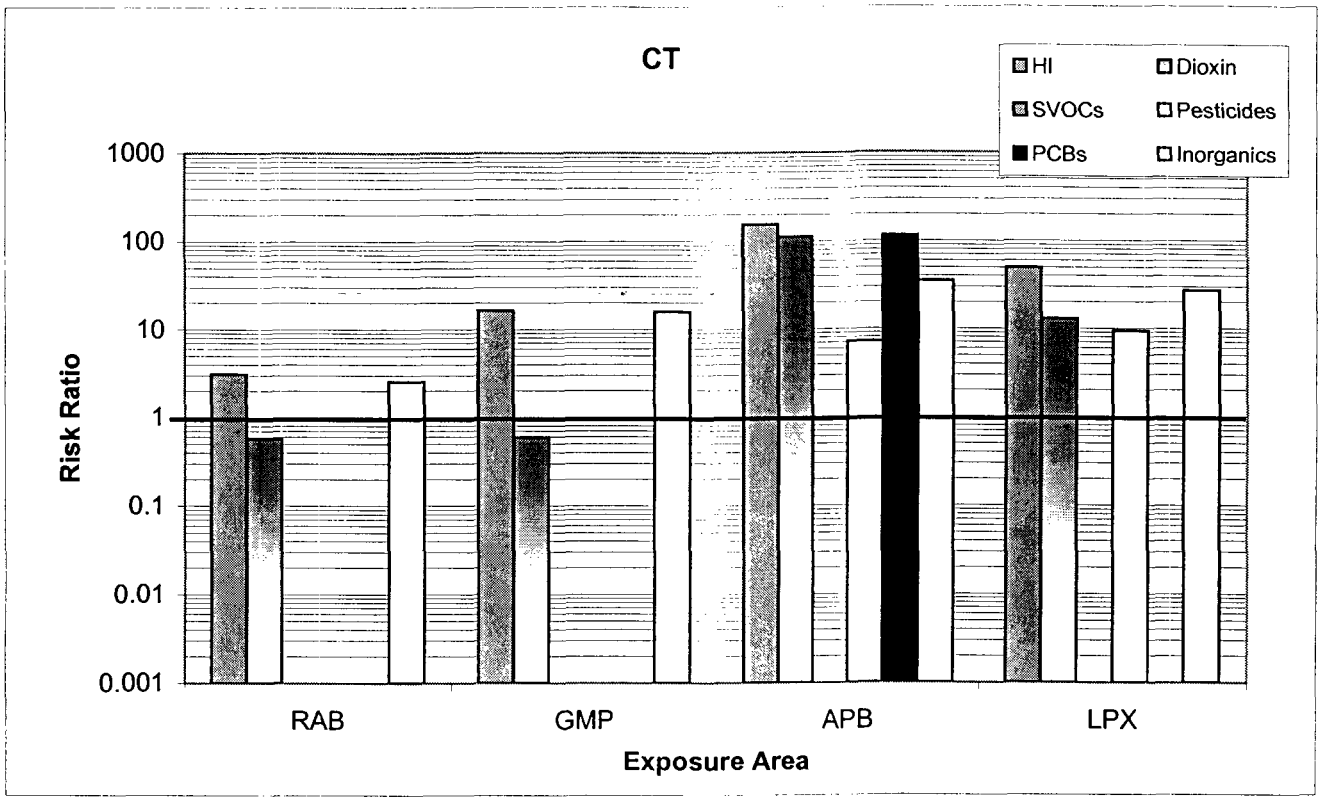
FIGURE 58
FIELD STUDY SAMPLING LOCATIONS AND SUMMARY OF MACROINVERTEBRATE BIOASSAY
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

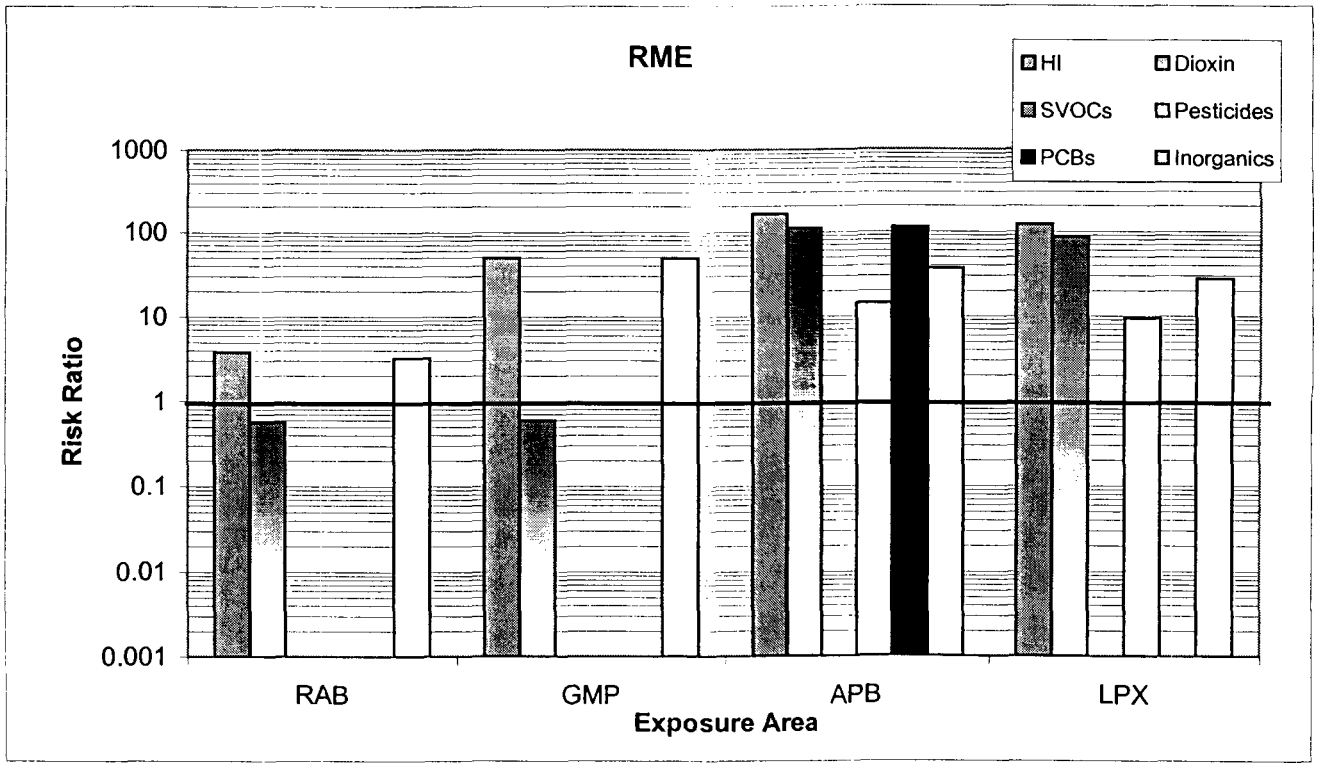
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centerdale_2003.dwg	BGF	51226.24	NAR	1/8/03	7/19/04

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FIGURE 59
RISK RATIOS BY EXPOSURE AREA FOR SURFACE WATER SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



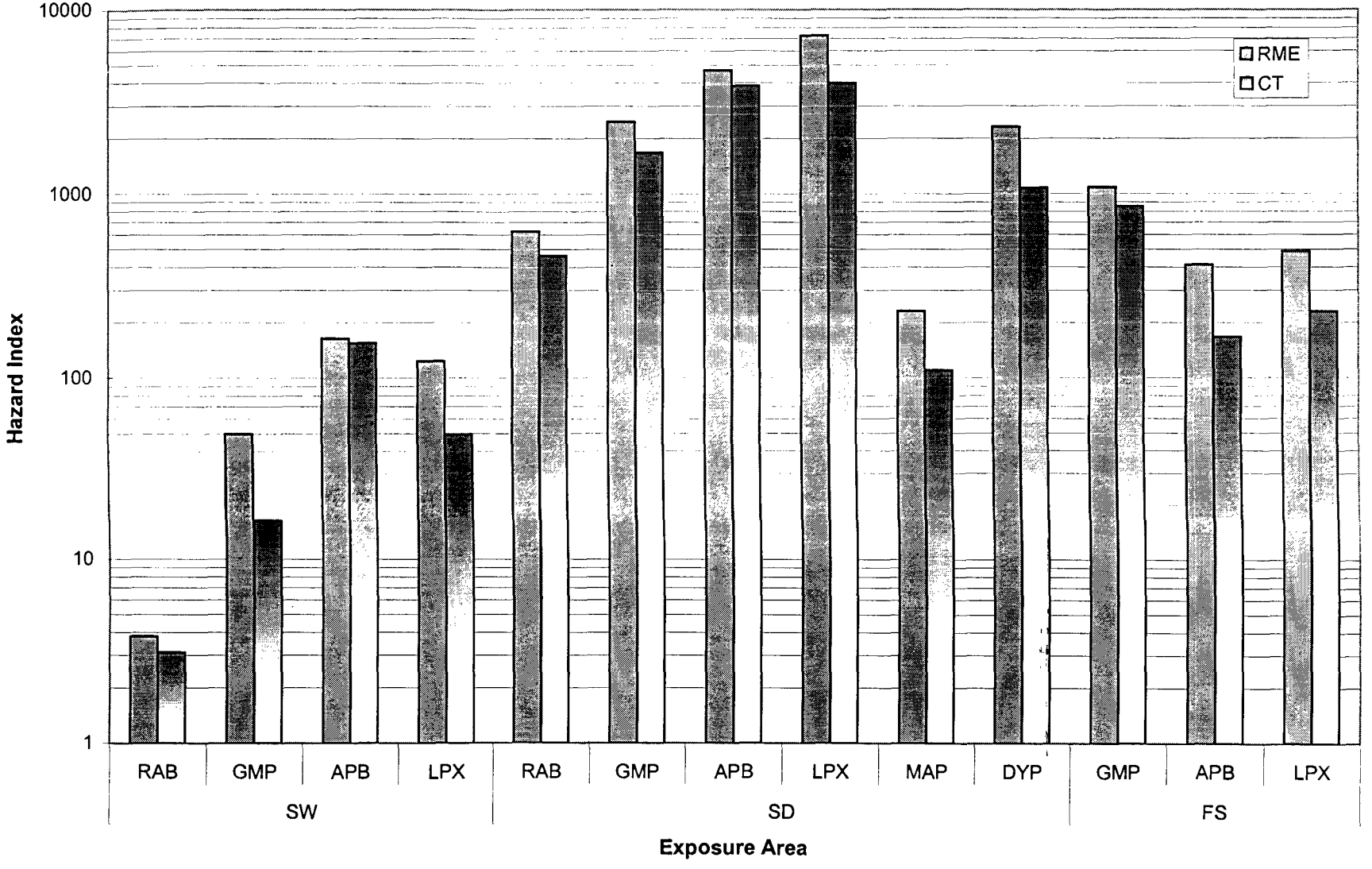


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FIGURE 60
RISK RATIOS BY EXPOSURE AREA FOR ABIOTIC MEDIA

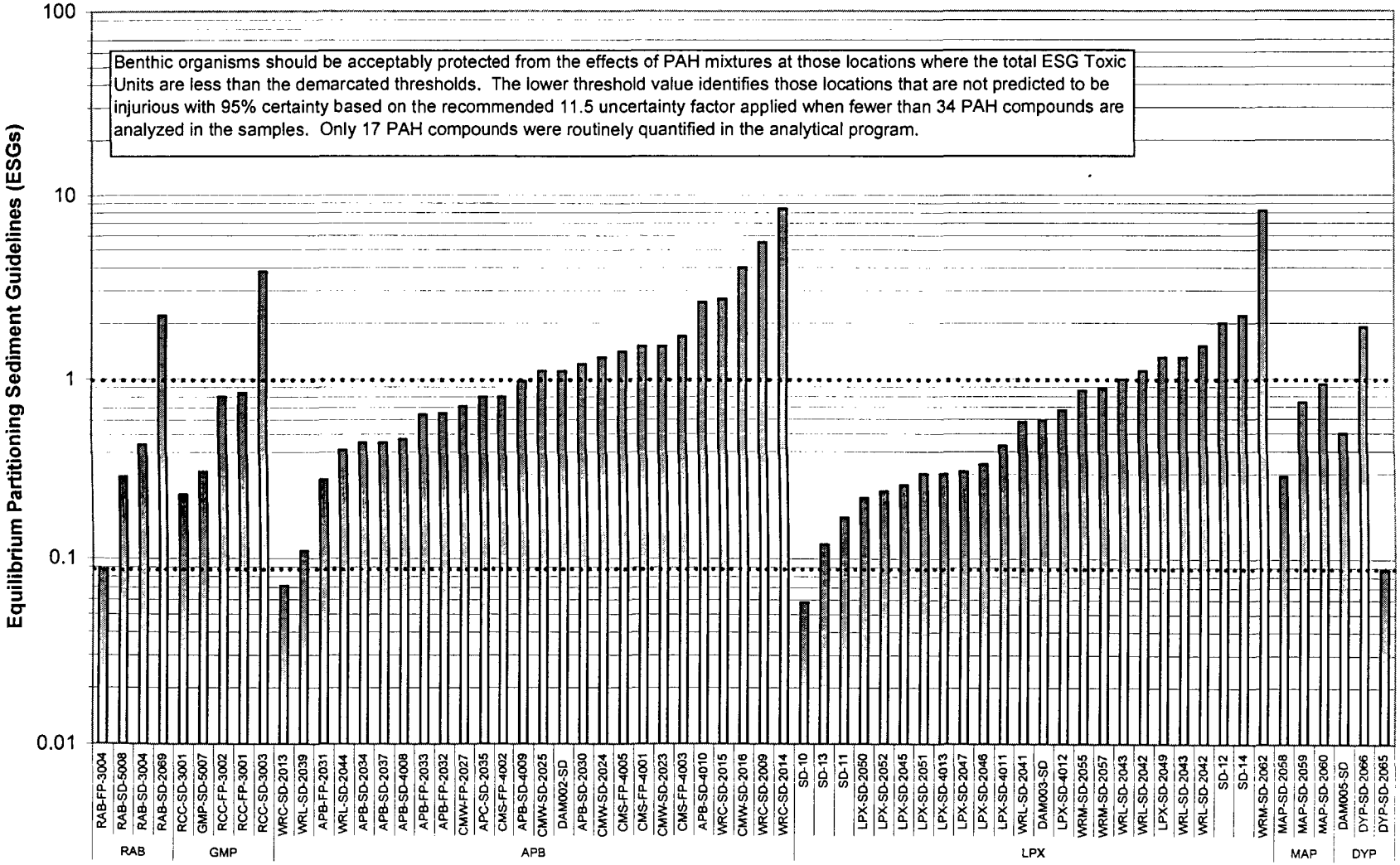
Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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FIGURE 62
EQUILIBRIUM PARTITIONING SEDIMENT GUIDELINES FOR PAH MIXTURES IN SEDIMENT

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



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FIGURE 63
CARBON-NORMALIZED EXCESS SEM IN SURFICIAL SEDIMENT

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

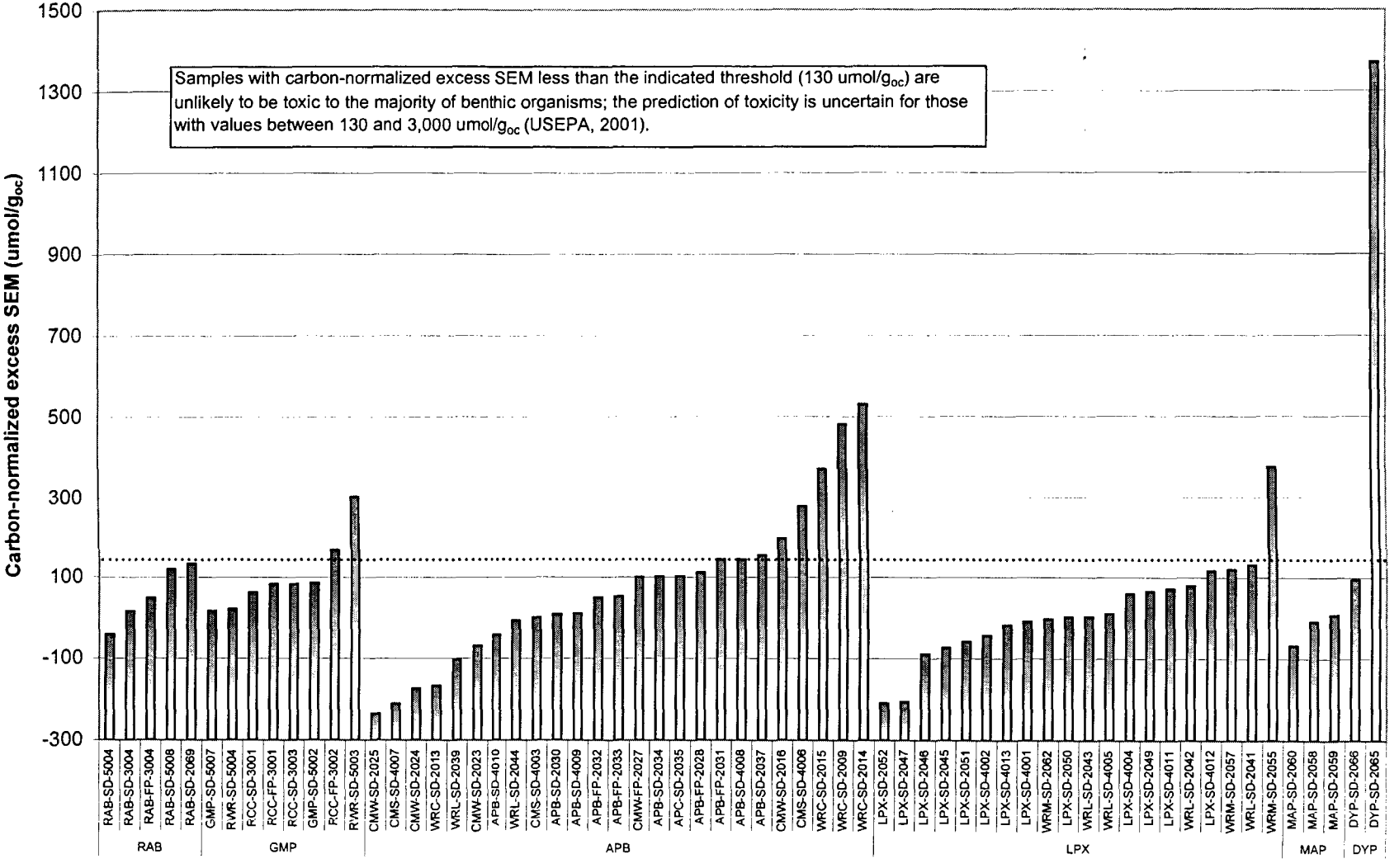
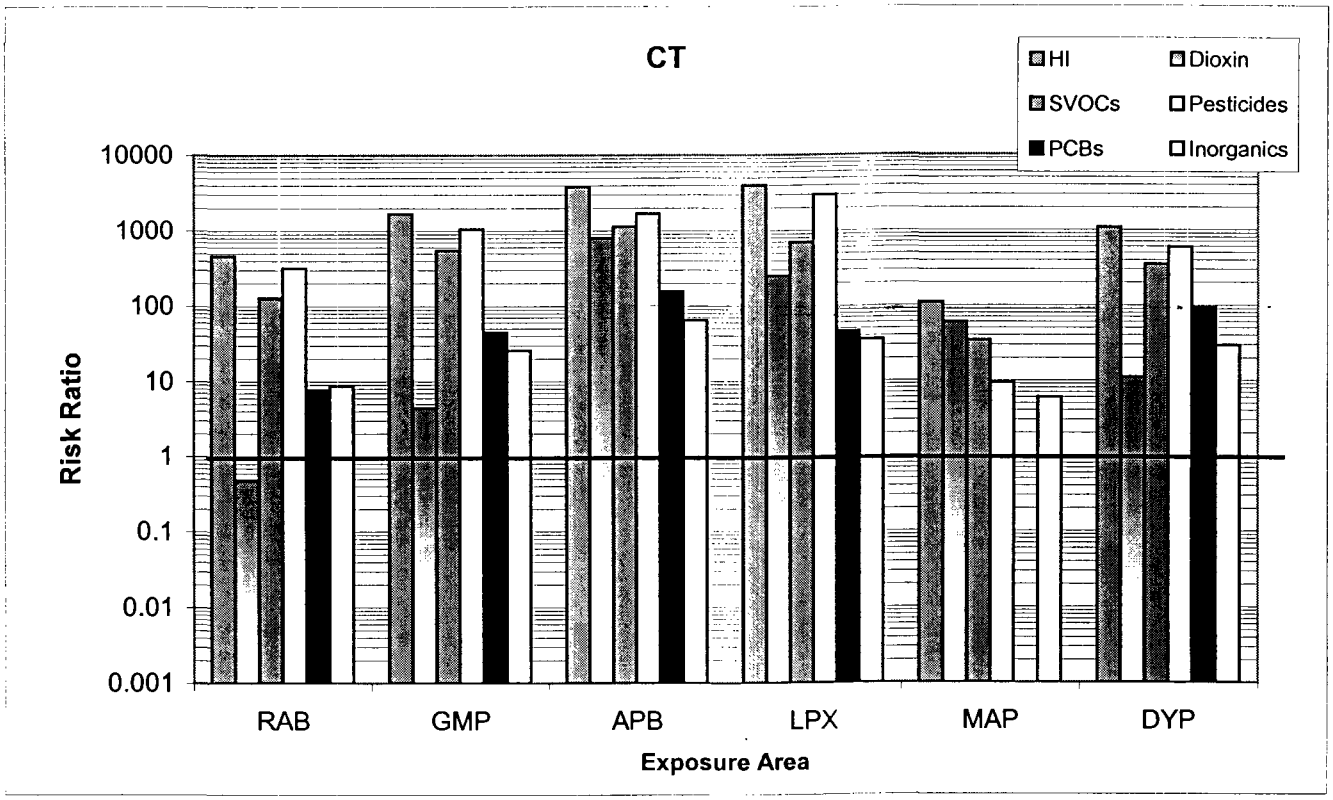
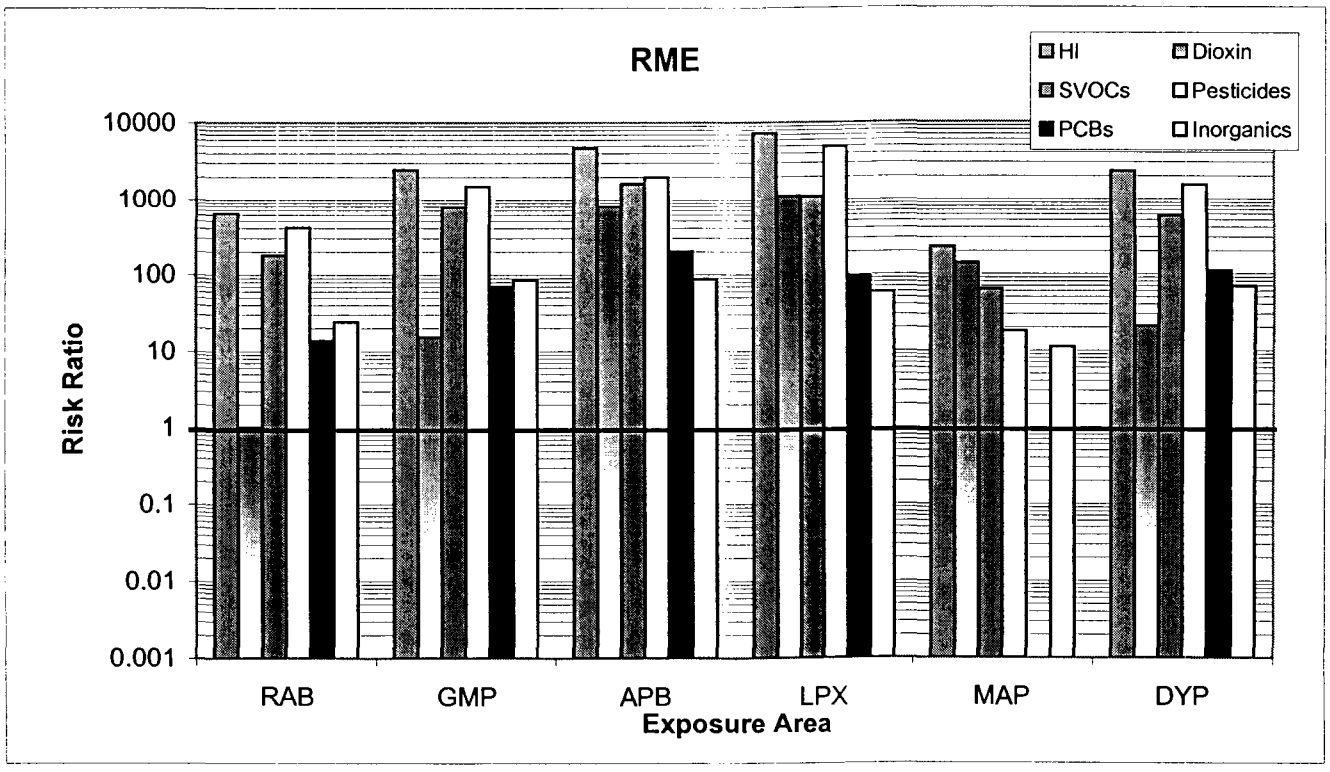


FIGURE 61
RISK RATIOS BY EXPOSURE AREA FOR SEDIMENT SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

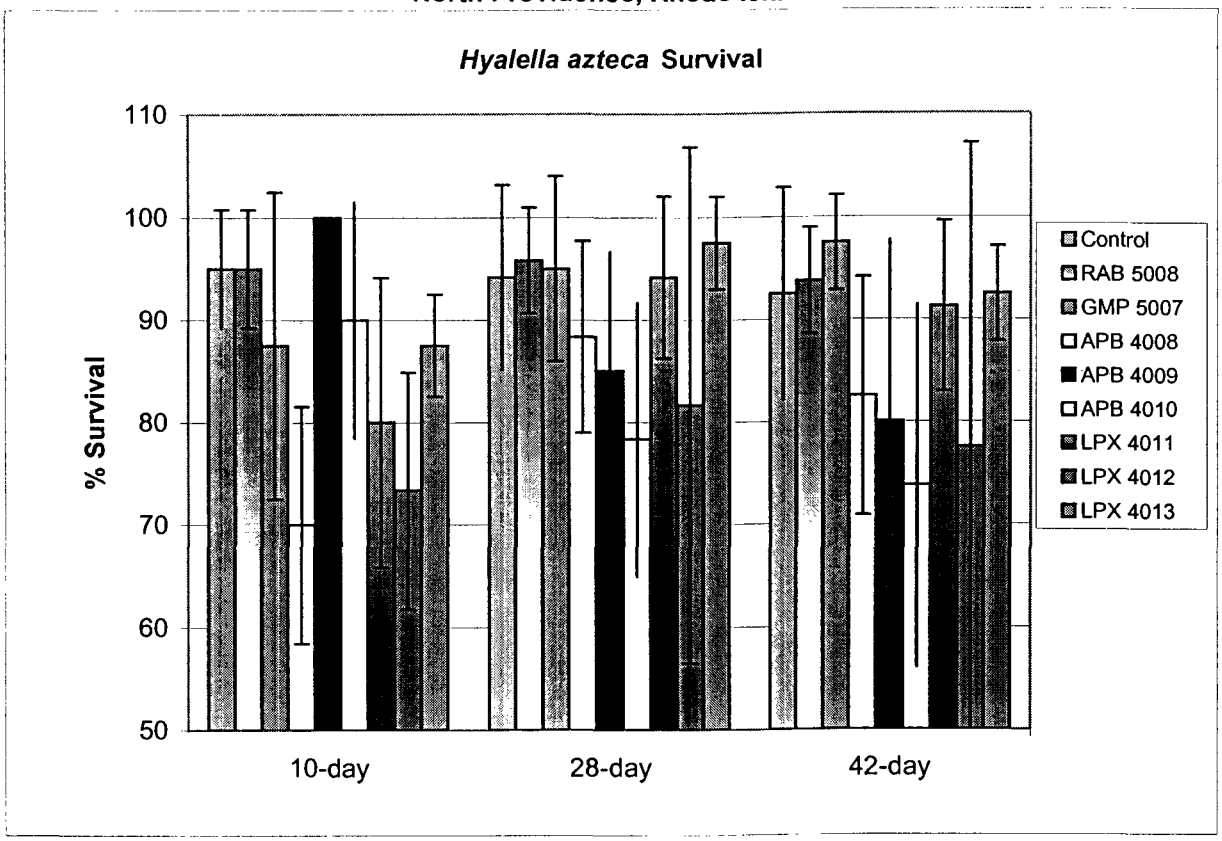


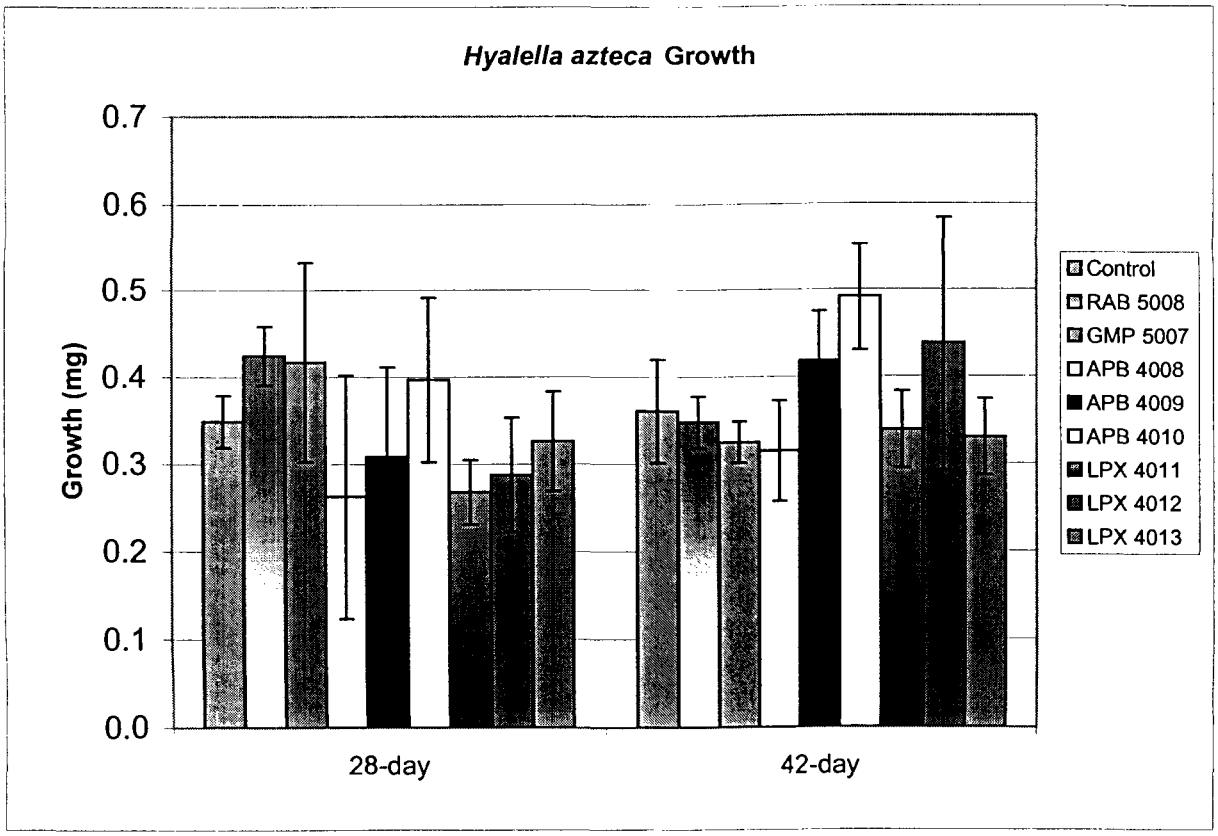


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FIGURE 64
SUMMARY OF AMPHIPOD MORTALITY AND GROWTH EFFECTS DATA

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





MACTEC Engineering and Consulting, Inc.
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Bioassay Results(JUL04), Figure 3

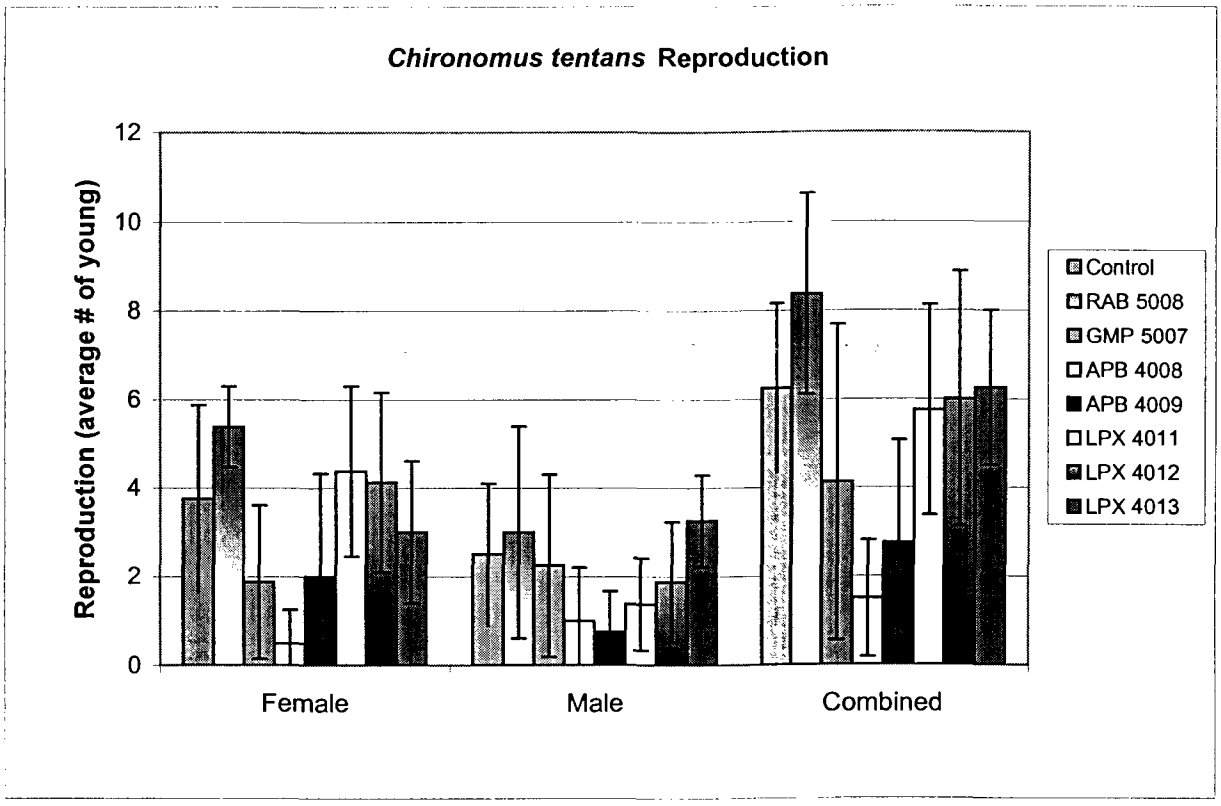
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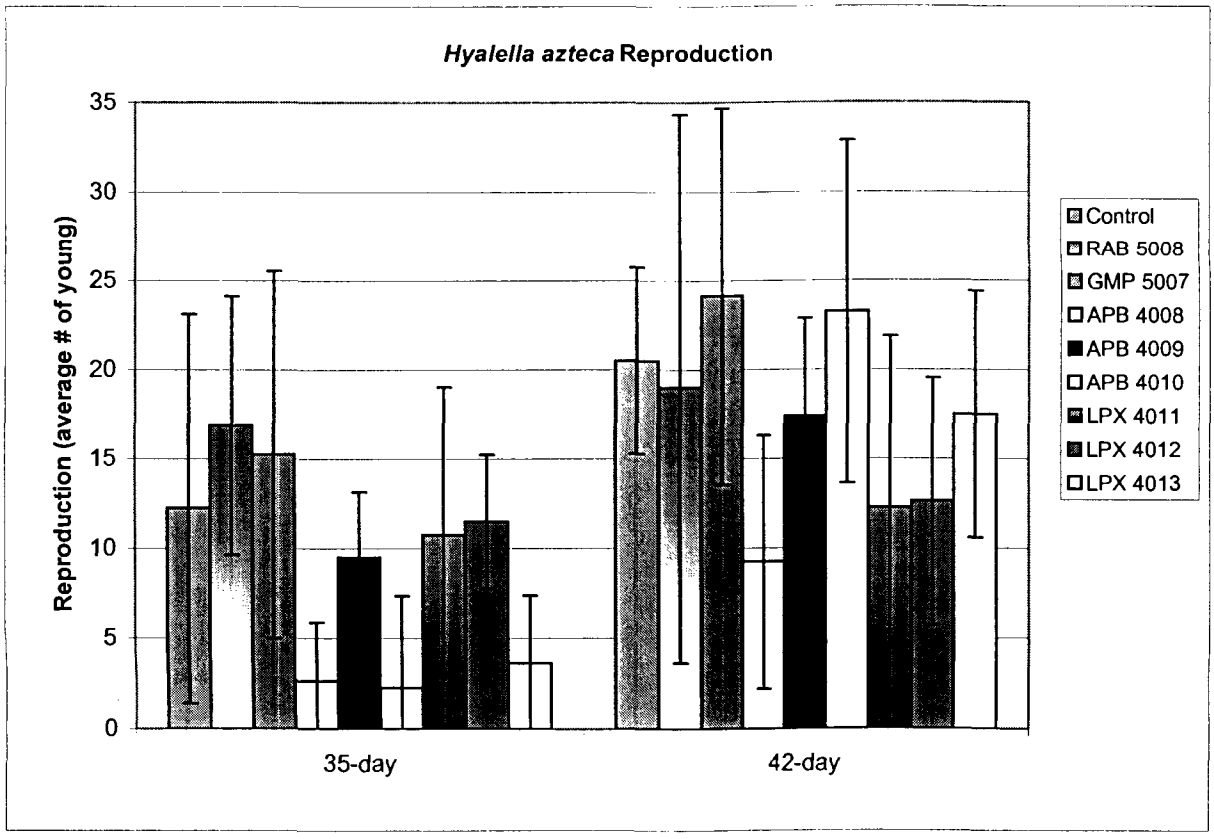
US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

FIGURE 65
SUMMARY OF MIDGE AND AMPHIPOD REPRODUCTIVE EFFECT DATA

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

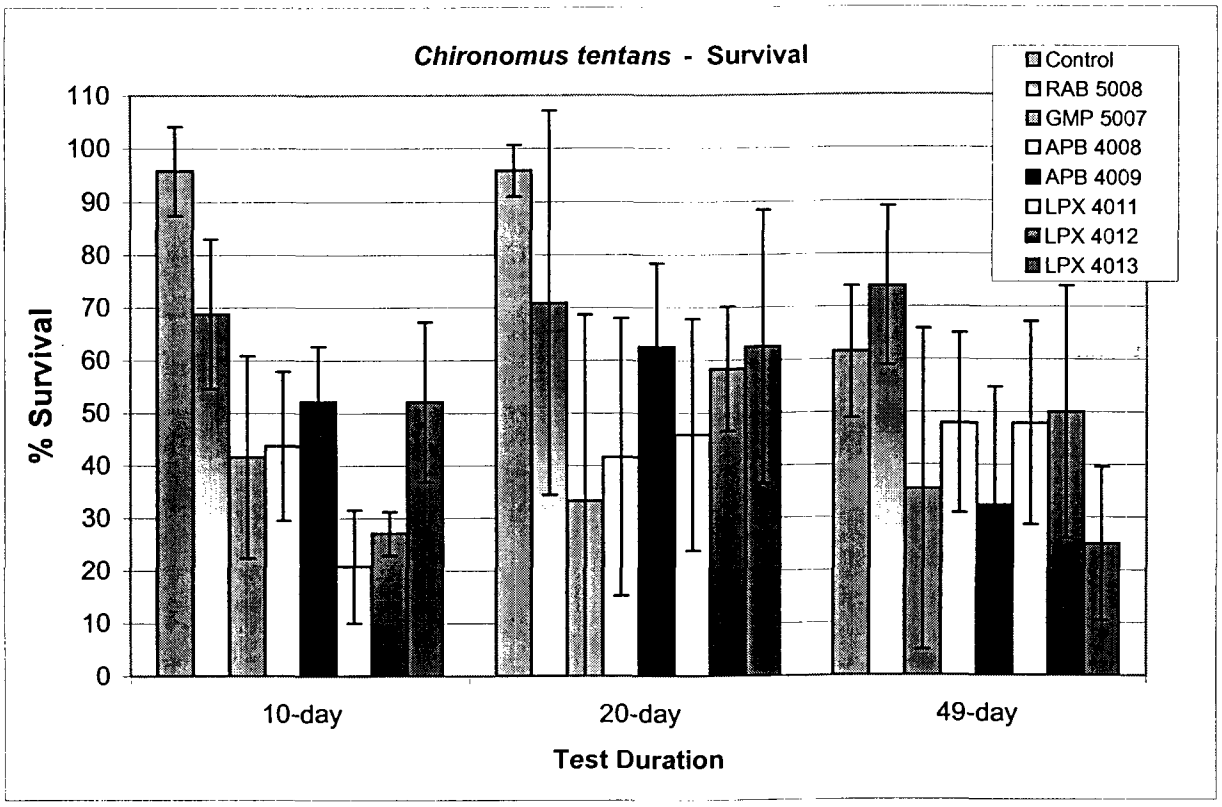


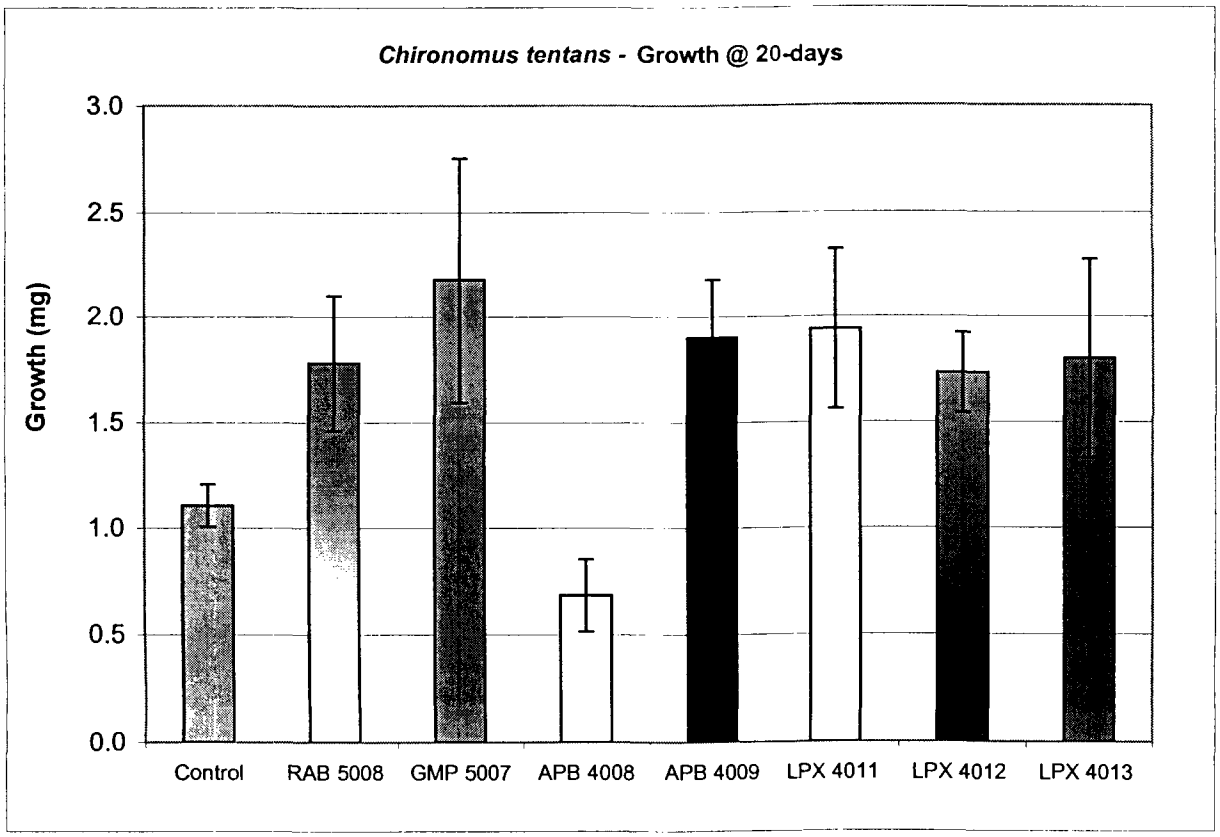


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FIGURE 66
SUMMARY OF MIDGE MORTALITY AND GROWTH EFFECTS DATA

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

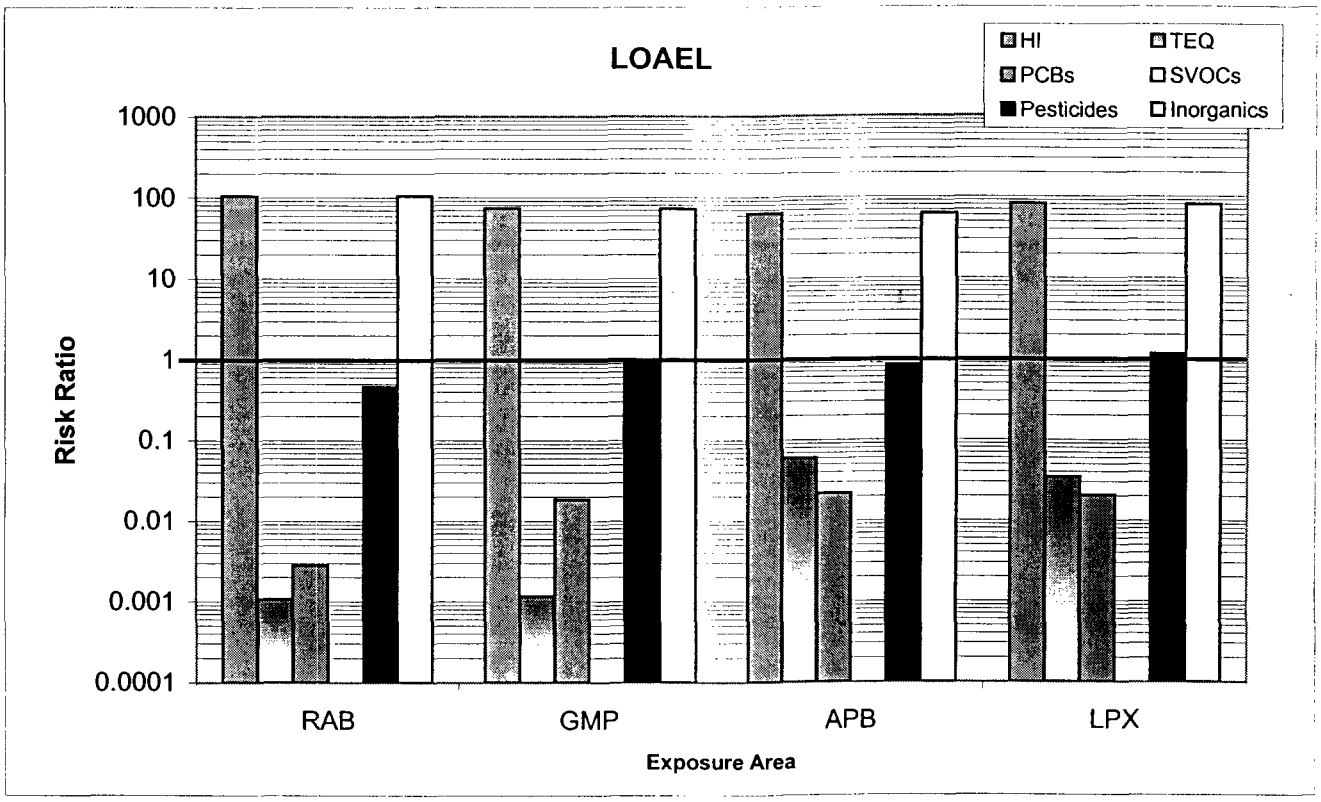




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FIGURE 67
CBR RISK RATIOS BY EXPOSURE AREA FOR CRAYFISH SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



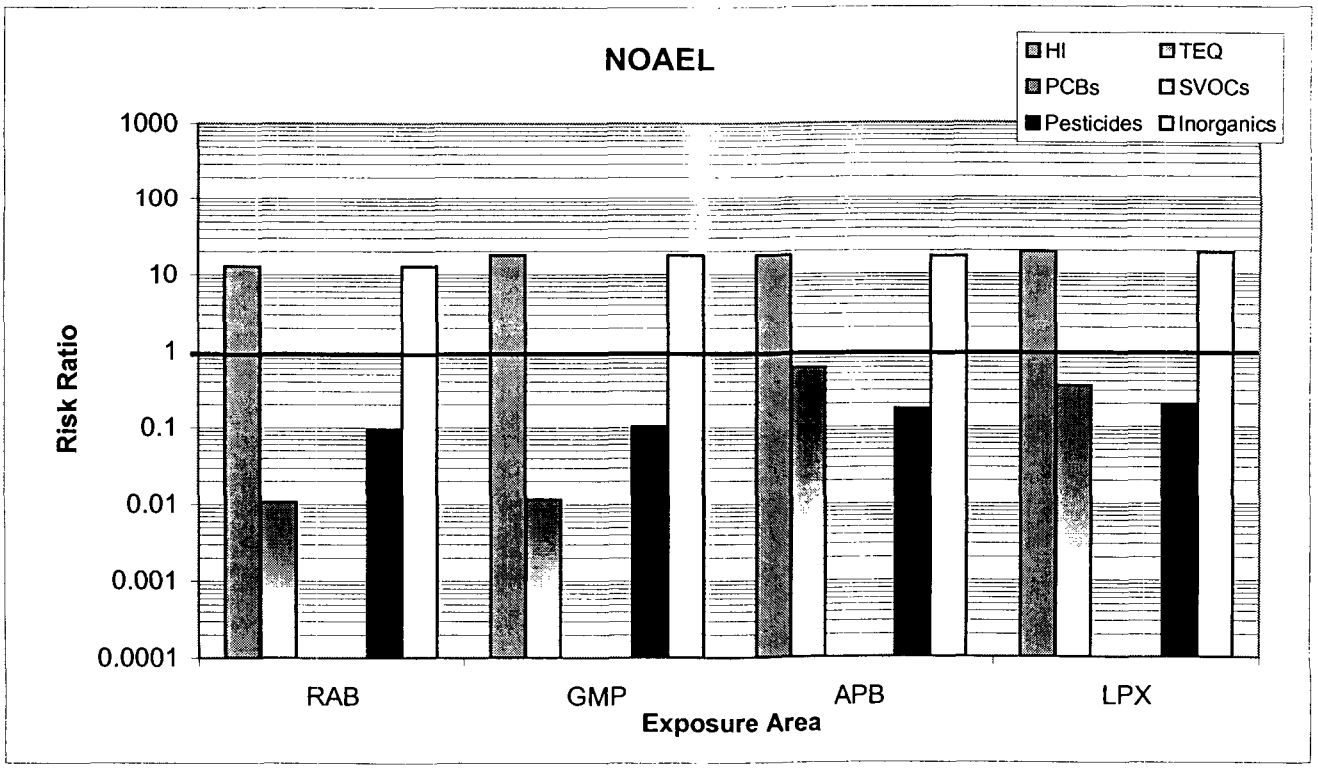
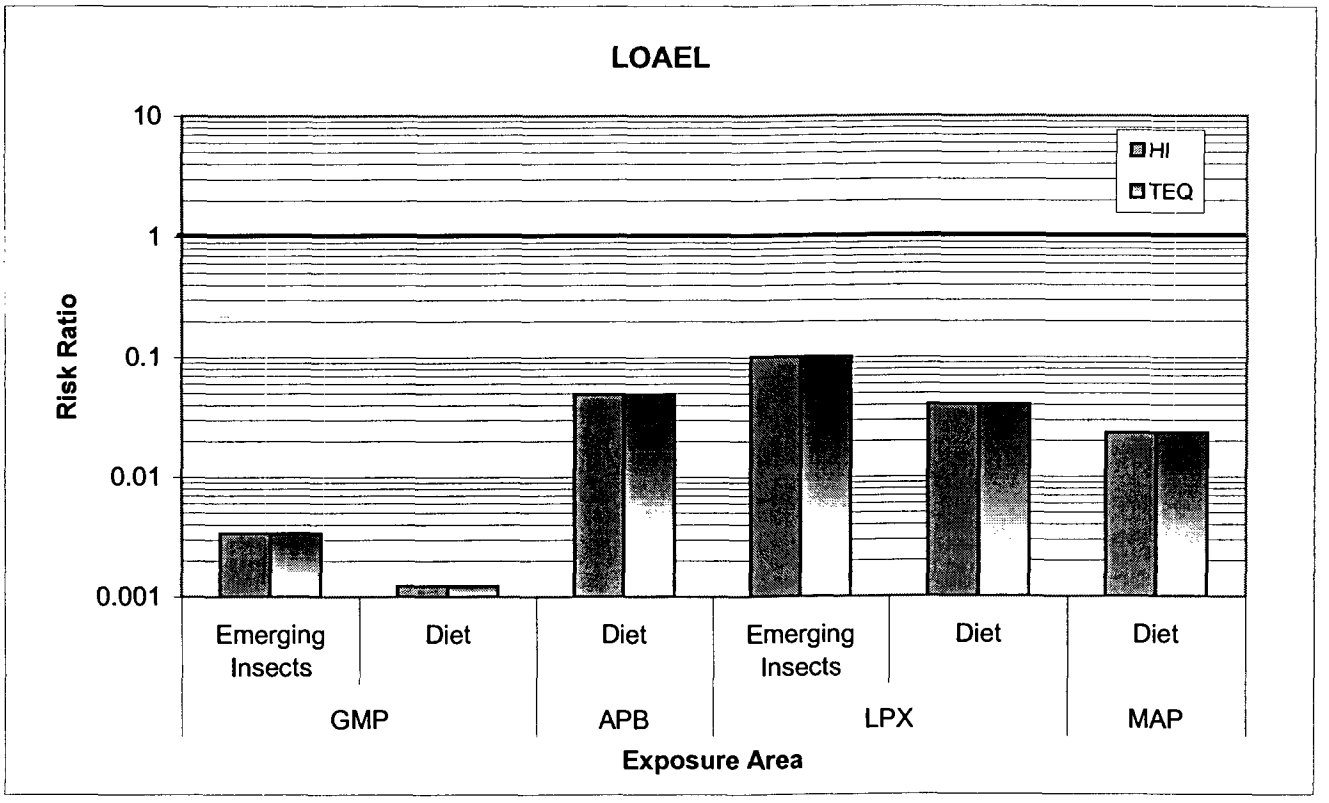
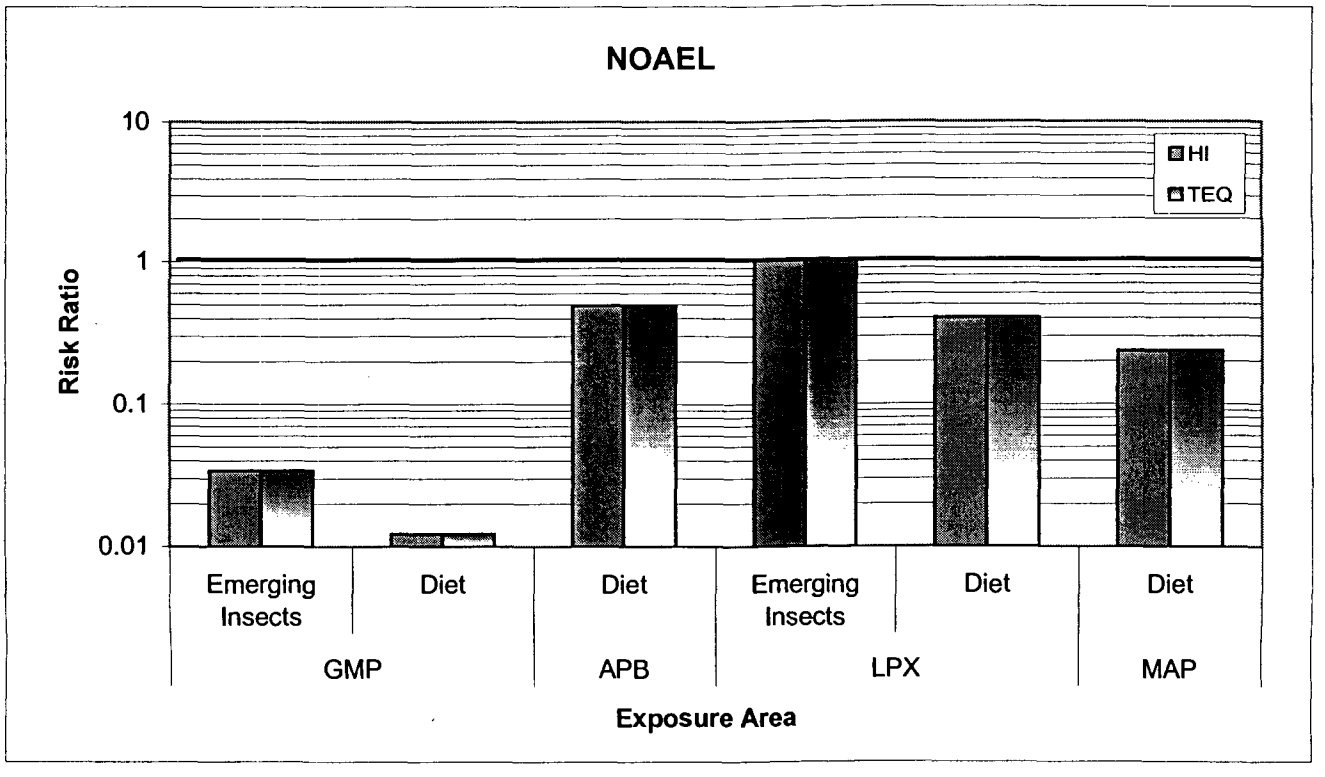


FIGURE 68
CBR RISK RATIOS BY EXPOSURE AREA FOR EMERGING INSECTS/SWALLOW DIET SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

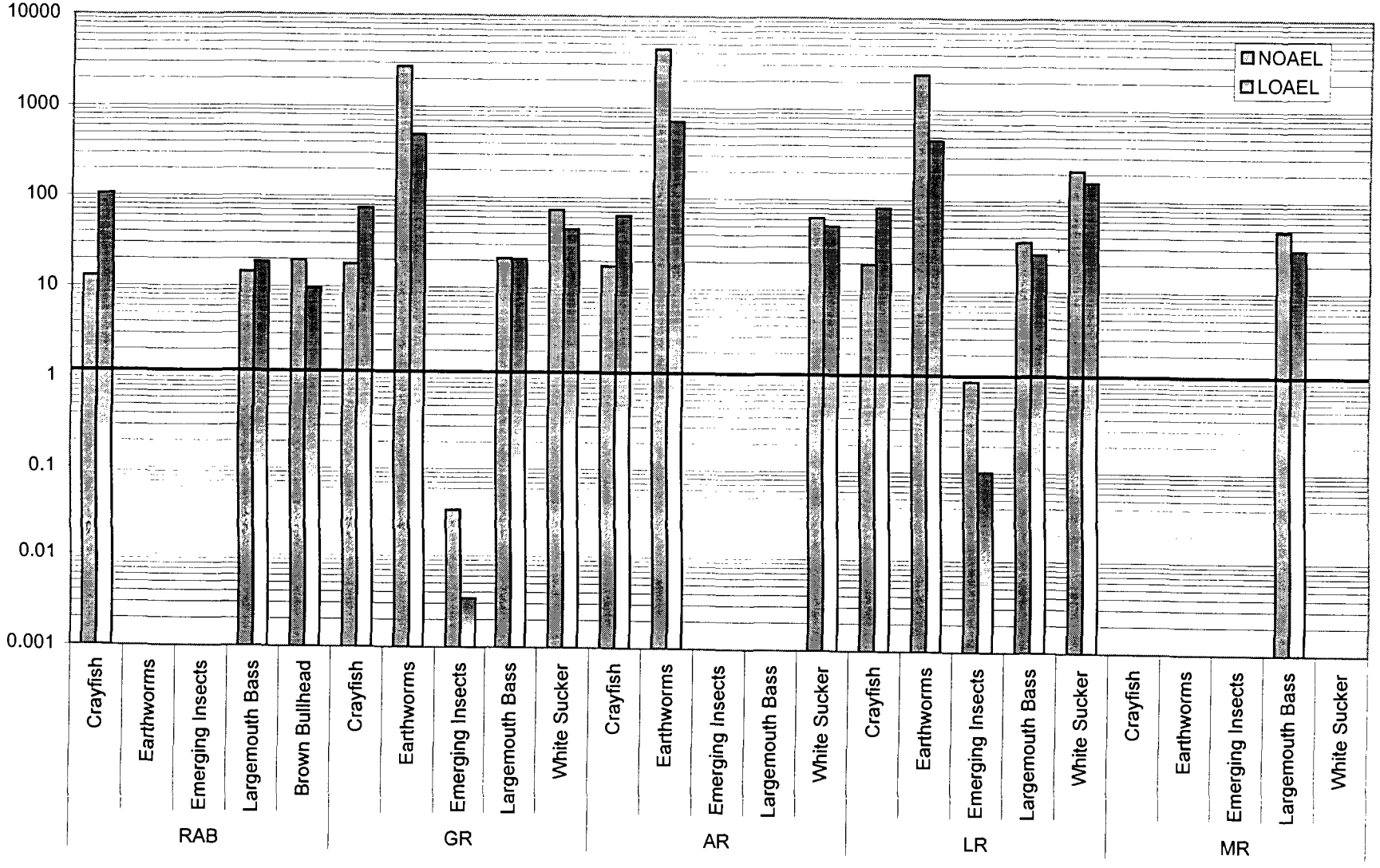




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FIGURE 69
SUMMARY OF CBR RISK RATIOS FOR AQUATIC BIOTA TISSUE

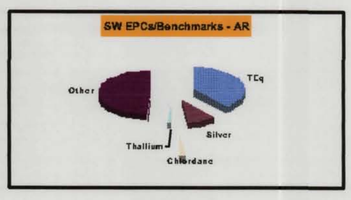
Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



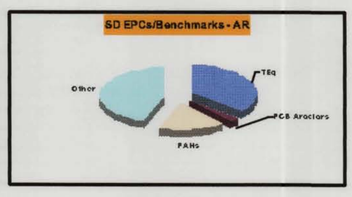
MACTEC Engineering and Consulting, Inc.
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Biota_EPCS - Final(rev), CBR Risk Summary - Area

8/4/2004

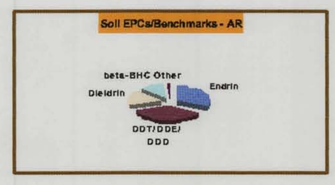
1. SW EPCs/Benchmarks (AR)			
Analyte	RME	CT	%
TEq	1.1E+02	1.1E+02	39%
Silver	2.6E+01	2.6E+01	9%
Chlordane	5.8E+00	5.8E+00	2%
Thallium	2.9E+00	2.9E+00	1%
Other	1.5E+02	1.4E+02	49%
Incremental HI	**2.9E+02**	**2.9E+02**	



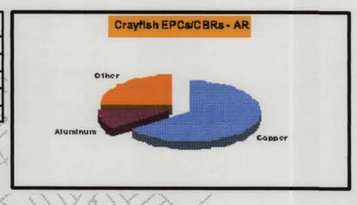
2b. SD EPCs/Benchmarks (AR)			
Analyte	RME	CT	%
TEq	7.8E+02	8.0E+02	36%
PCB Aroclor	9.2E+02	5.8E+01	3%
PAHs	6.4E+02	3.7E+02	17%
Other	7.8E+02	9.7E+02	44%
Incremental HI	**2.3E+03**	**2.2E+03**	



2b. Floodplain Soil EPCs/Benchmarks (AR)			
Analyte	RME	CT	%
Endrin	1.1E+01	8.7E+00	32%
DDT/DDE/DDD	6.0E+01	1.0E+01	38%
Dieldrin	3.5E+01	4.6E+00	17%
beta-BHC	3.1E+00	3.1E+00	11%
Other	1.1E+00	3.0E-01	1%
Incremental HI	**1.1E+02**	**2.7E+01**	



4. Crayfish EPCs/CBRs (AR)			
Analyte	NOAEL	LOAEL	%
Copper	1.0E+00	8.0E-01	65%
Aluminum	6.7E-01	8.3E-02	7%
Other	7.7E-01	3.5E-01	28%
Incremental HI	**2.5E+00**	**1.2E+00**	



5. Earthworm Tissue/CBRs (AR)			
Analyte	NOAEL	LOAEL	%
Aluminum	1.6E+03	2.0E+02	89%
Barium	-	2.0E+01	9%
Other	2.2E+00	5.6E+00	2%
Incremental HI	**1.6E+03**	**2.3E+02**	



ALLENDALE REACH

ALLENDALE POND

ALLENDALE DAM

LYMAN MILL REACH

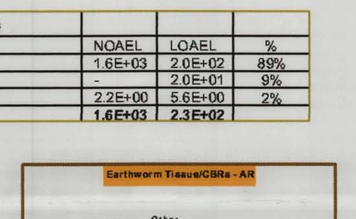
1. SW EPCs/Benchmarks (LR)			
Analyte	RME	CT	%
Silver	1.8E+01	1.8E+01	32%
TEq	8.5E+01	1.2E+01	22%
Chlordane	9.3E+00	9.3E+00	17%
Thallium	2.9E+00	2.9E+00	5%
Other	1.2E+01	1.3E+01	23%
Incremental HI	**1.3E+02**	**5.6E+01**	



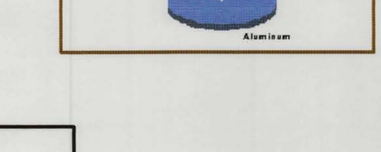
2b. SD EPCs/Benchmarks (LR)			
Analyte	RME	CT	%
Technical Chlordane	3.2E+03	1.8E+03	71%
TEq	1.1E+03	2.4E+02	10%
PAHs	1.6E+02	7.6E+01	3%
Dieldrin	9.3E+01	6.3E+01	2%
Other	4.5E+02	3.6E+02	14%
Incremental HI	**5.0E+03**	**2.5E+03**	



2b. Floodplain Soil EPCs/Benchmarks (LR)			
Analyte	RME	CT	%
DDT/DDE	1.4E+01	5.2E+00	71%
Zinc	6.5E+00	2.0E+00	27%
Other	2.5E+00	1.3E-01	2%
Incremental HI	**2.3E+01**	**7.4E+00**	



4. Crayfish EPCs/CBRs (LR)			
Analyte	NOAEL	LOAEL	%
Barium		6.3E+00	76%
Copper	9.6E-01	7.4E-01	9%
Other	1.7E+00	1.2E+00	15%
Incremental HI	**2.7E+00**	**8.2E+00**	



5. Earthworm Tissue/CBRs (LR)			
Analyte	NOAEL	LOAEL	%
Cadmium	-	2.5E+01	58%
Selenium	-	7.2E+00	17%
Lead	1.6E+01	6.1E+00	14%
Zinc	6.9E+00	4.5E+00	11%
Incremental HI	**2.3E+01**	**4.5E+01**	



ASSAPUMPSET POND

ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

LYMAN MILL POND

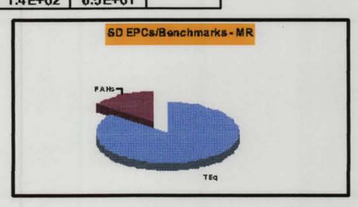
LYMAN MILL DAM

MANTON REACH

MANTON POND

MANTON DAM

2b. SD EPCs/Benchmarks (MR)			
Analyte	RME	CT	%
TEq	1.2E+02	5.6E+01	85%
PAHs	2.0E+01	9.8E+00	15%
Incremental HI	**1.4E+02**	**6.5E+01**	

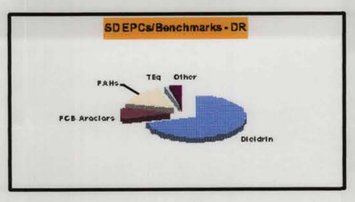


DYERVILLE REACH

DYERVILLE POND

DYERVILLE DAM

2b. SD EPCs/Benchmarks (DR)			
Analyte	RME	CT	%
Dieldrin	1.4E+03	4.5E+02	71%
PCB Aroclors	5.3E+01	5.2E+01	8%
PAHs	1.6E+02	9.8E+01	16%
TEq	5.9E+00	6.8E+00	1%
Other	5.3E+01	2.6E+01	4%
Incremental HI	**1.6E+03**	**6.3E+02**	



300 0 300 600

SCALE IN FEET

MACTEC
107 Audubon Road
Building II, #301
Wakefield, MA 01880
781-245-6606

FIGURE 70
INCREMENTAL RISK SUMMARY FOR AQUATIC AND FLOODPLAIN SOIL MACROINVERTEBRATE COMMUNITIES
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

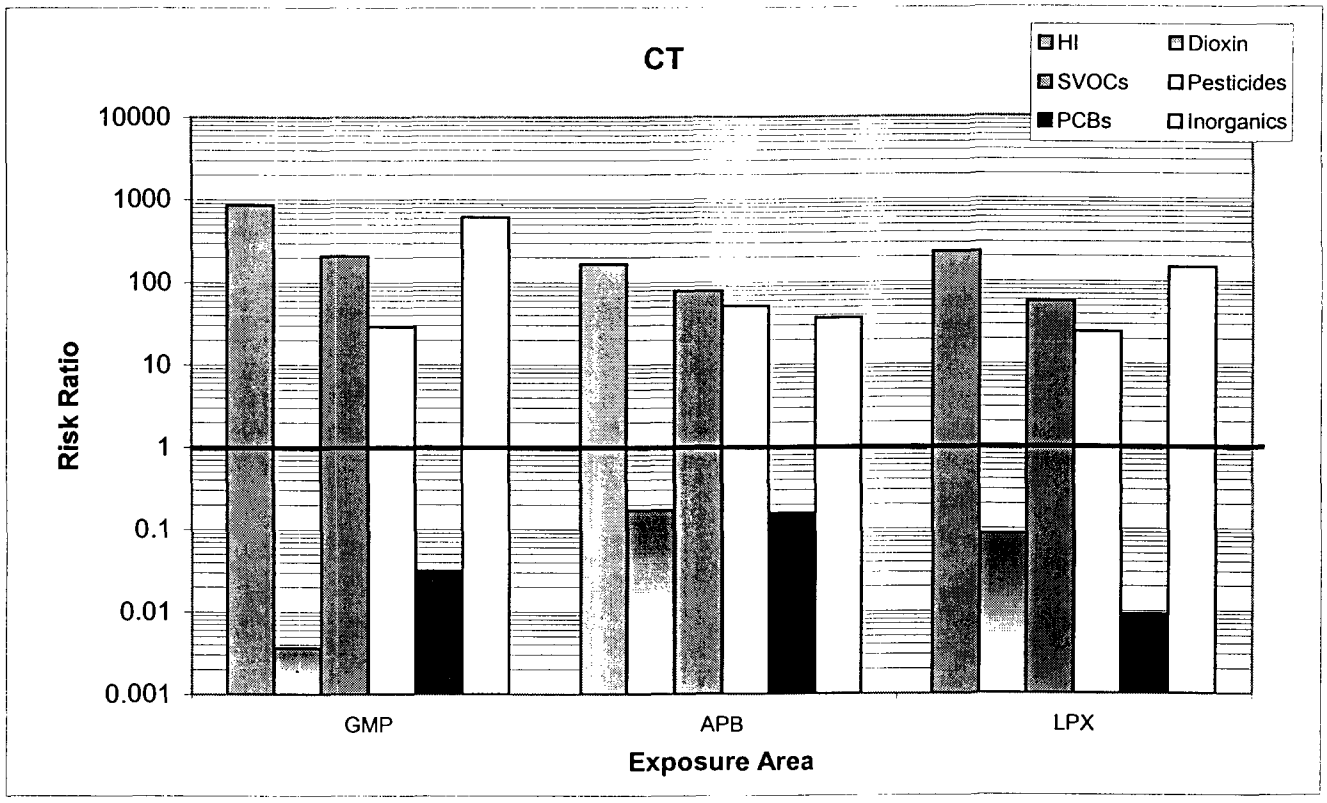
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centerdale2003a.dwg	BGF	51226.24	NAR	1/8/03	7/19/04

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FIGURE 71
RISK RATIOS BY EXPOSURE AREA FOR FLOODPLAIN SOIL SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



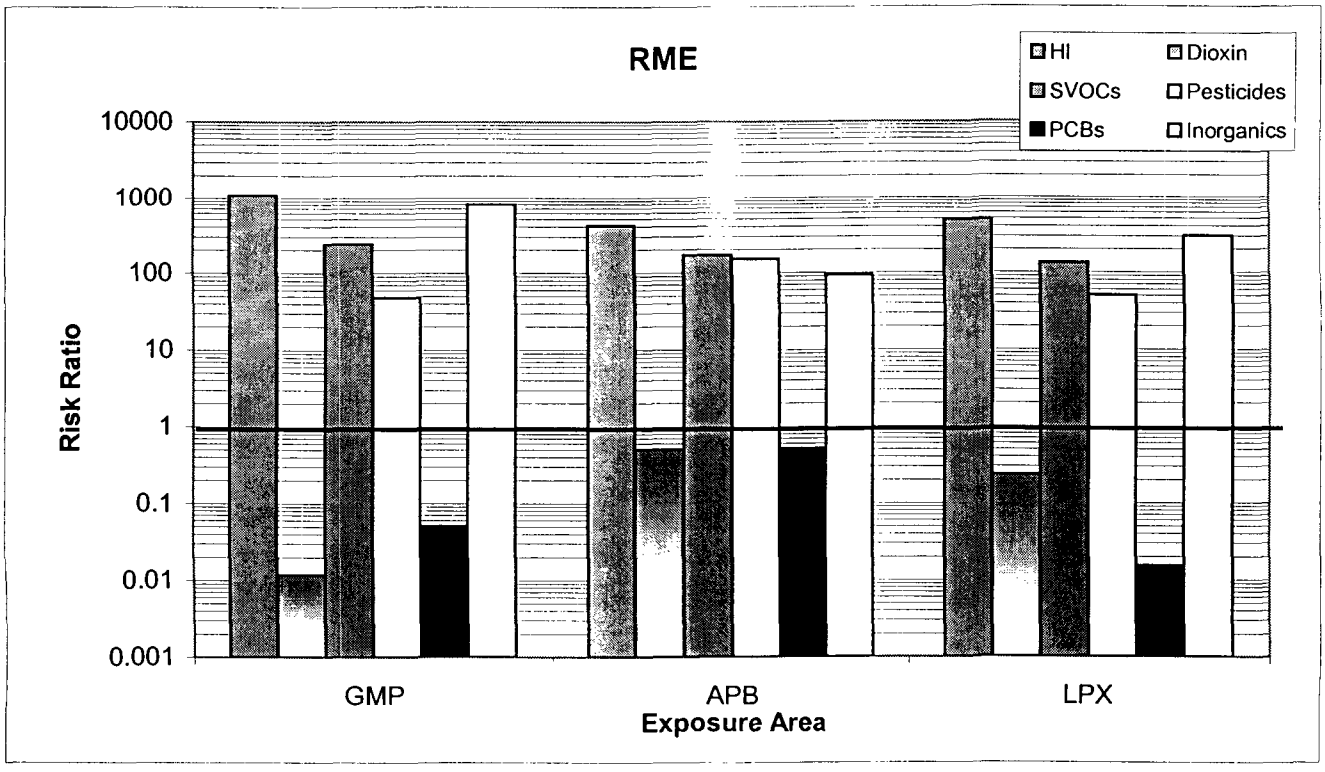
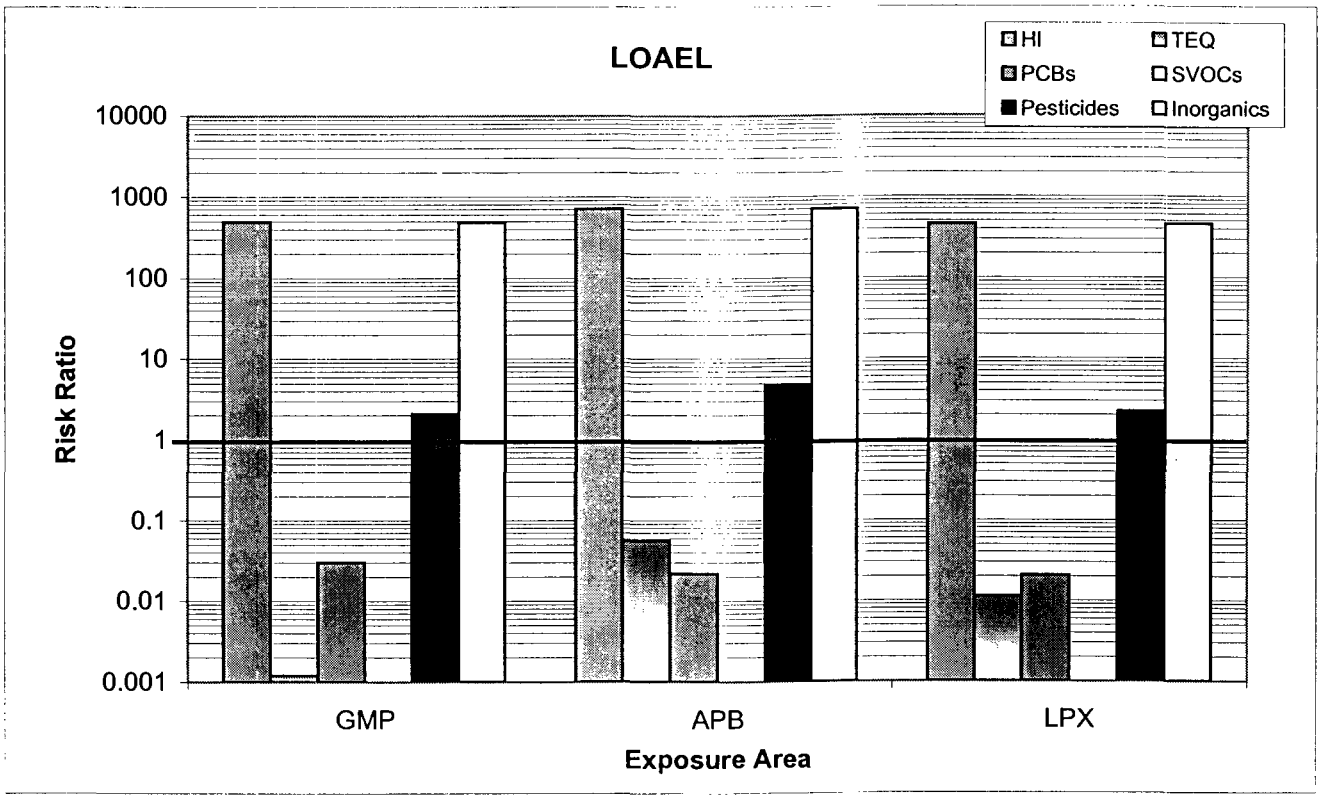
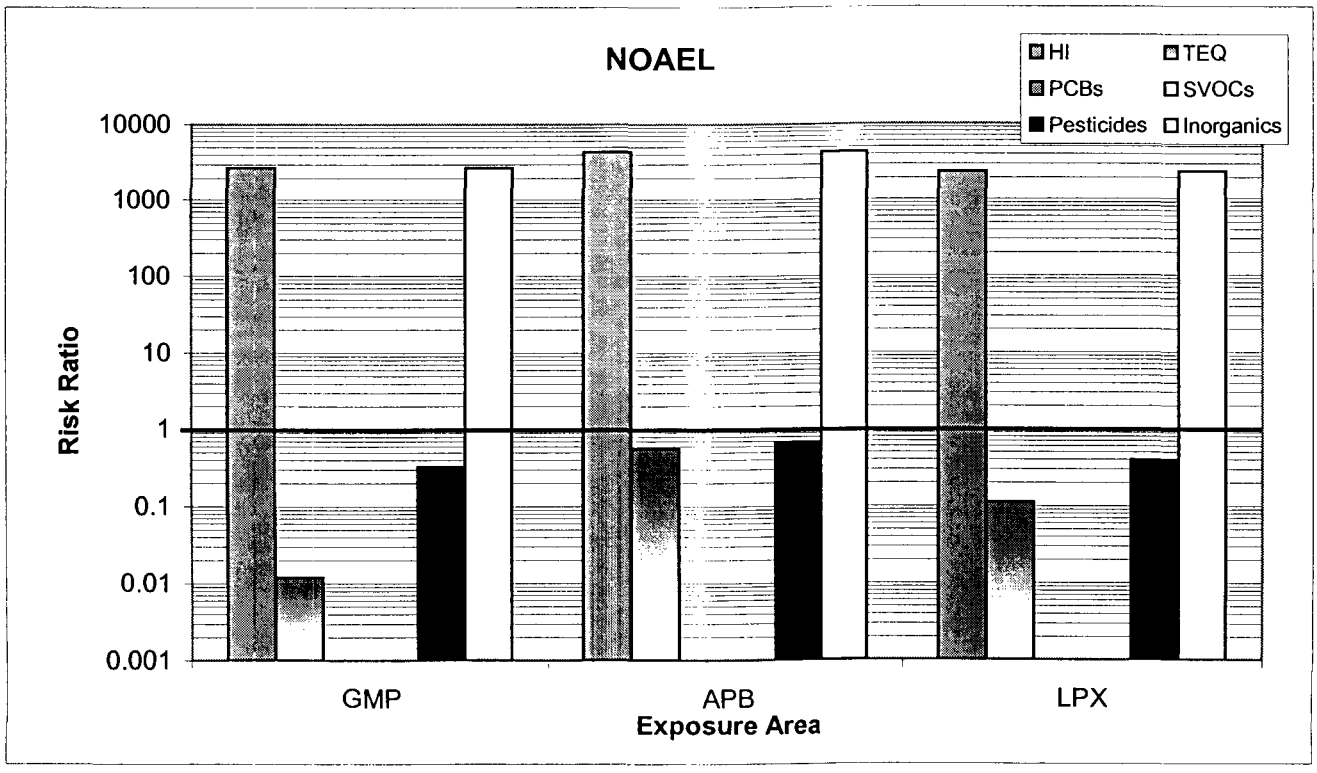


FIGURE 72
CBR RISK RATIOS BY EXPOSURE AREA FOR EARTHWORM SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

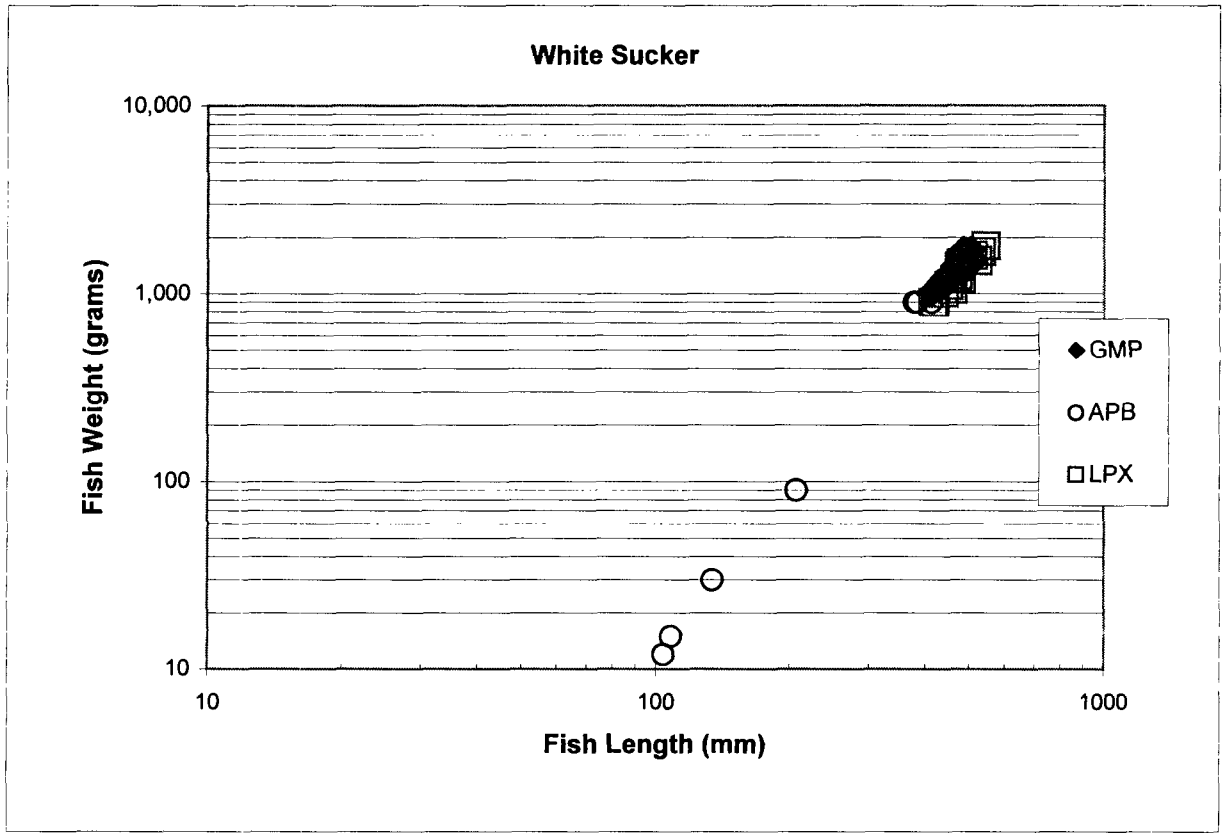


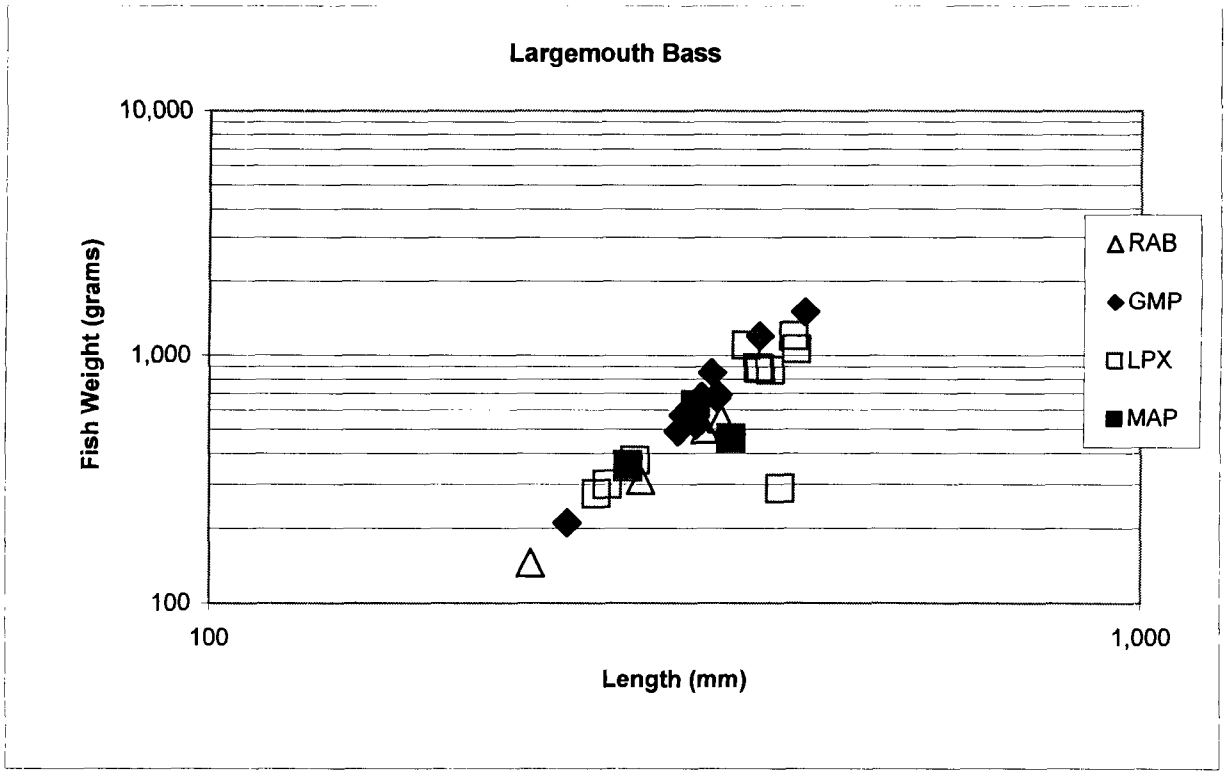


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FIGURE 73
FISH LENGTH-WEIGHT RELATIONSHIP FOR WHITE SUCKER AND LARGEMOUTH BASS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



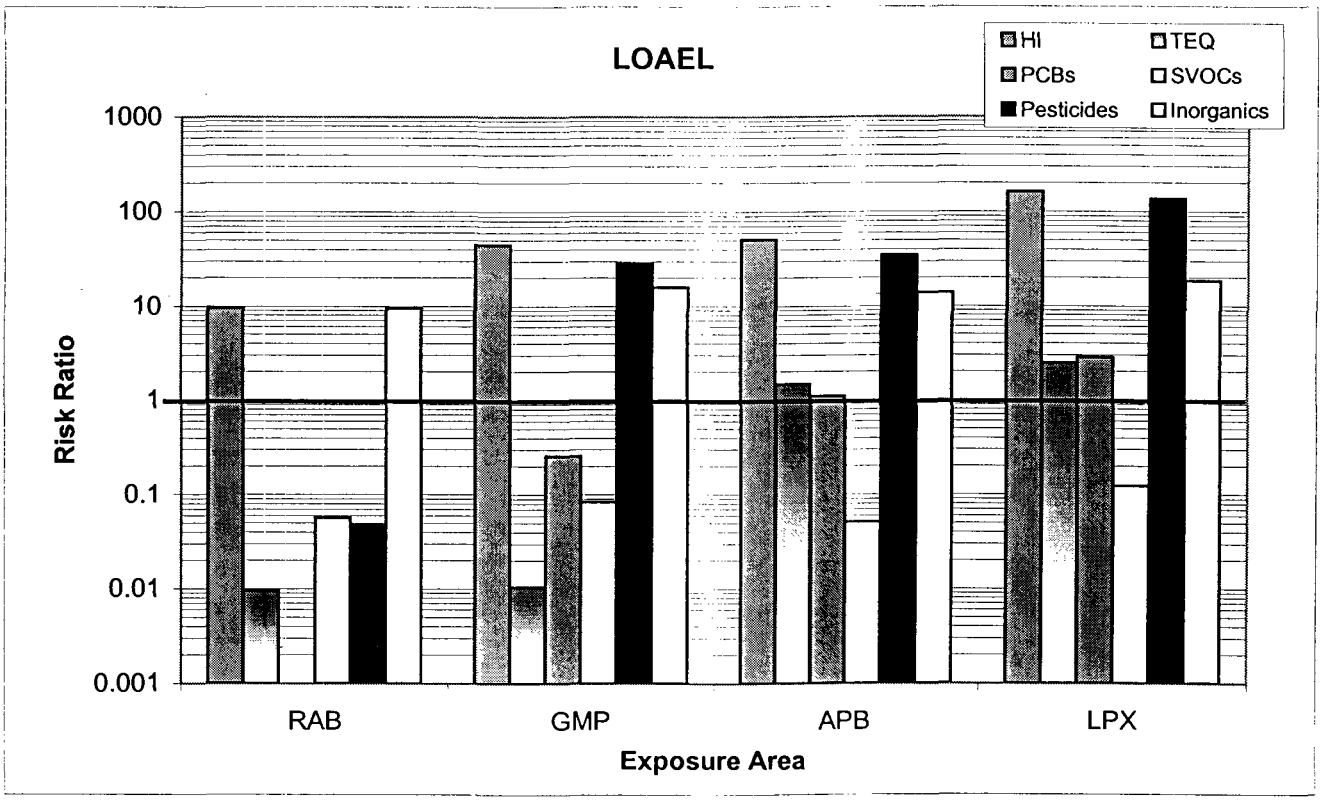


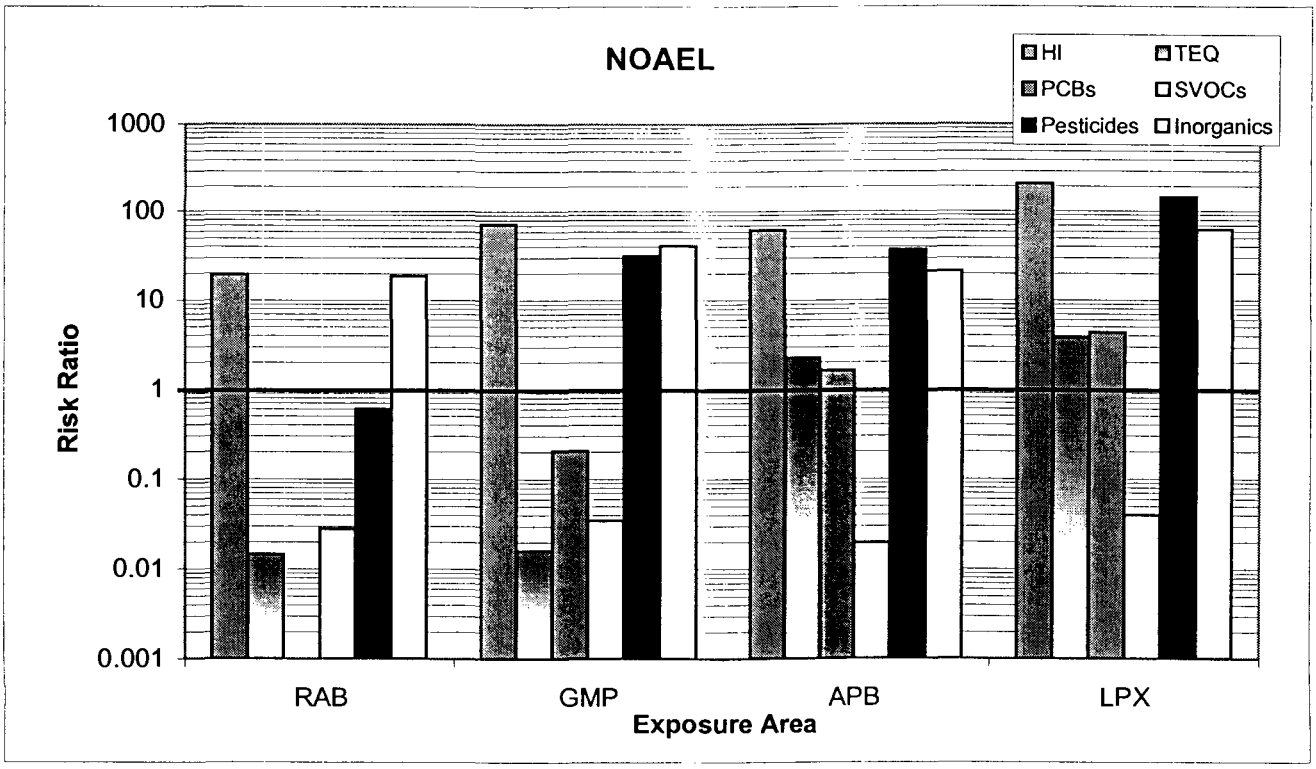
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Figures\
Fish length-weight(JUL04), Fish L-W

9/20/2004

FIGURE 74
CBR RISK RATIOS BY EXPOSURE AREA FOR WHITE SUCKER SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



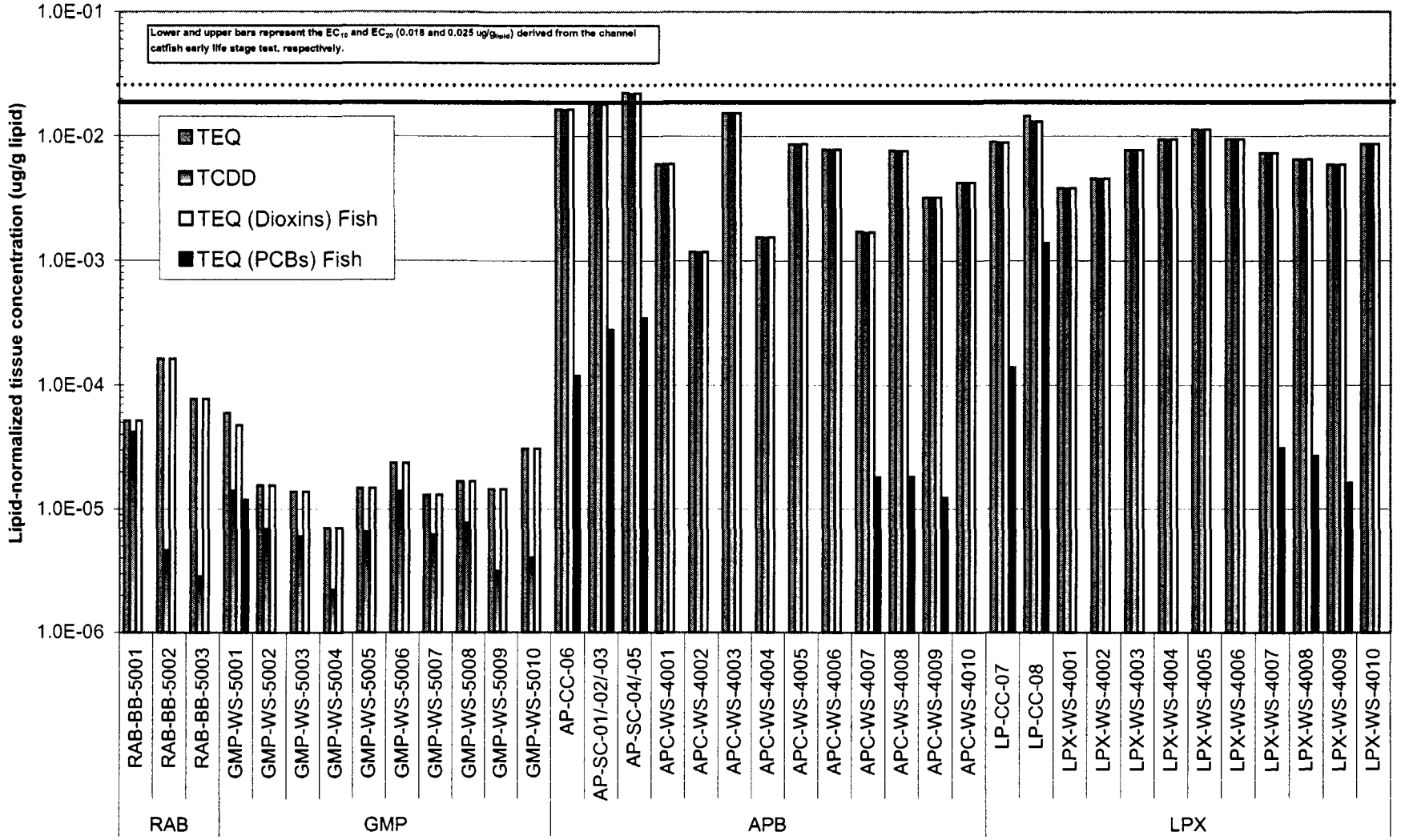


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FIGURE 75
COMPARISON OF TCDD AND TEQs IN DEMERSAL FISH TISSUE TO ELS EFFECT THRESHOLD CONCENTRATIONS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



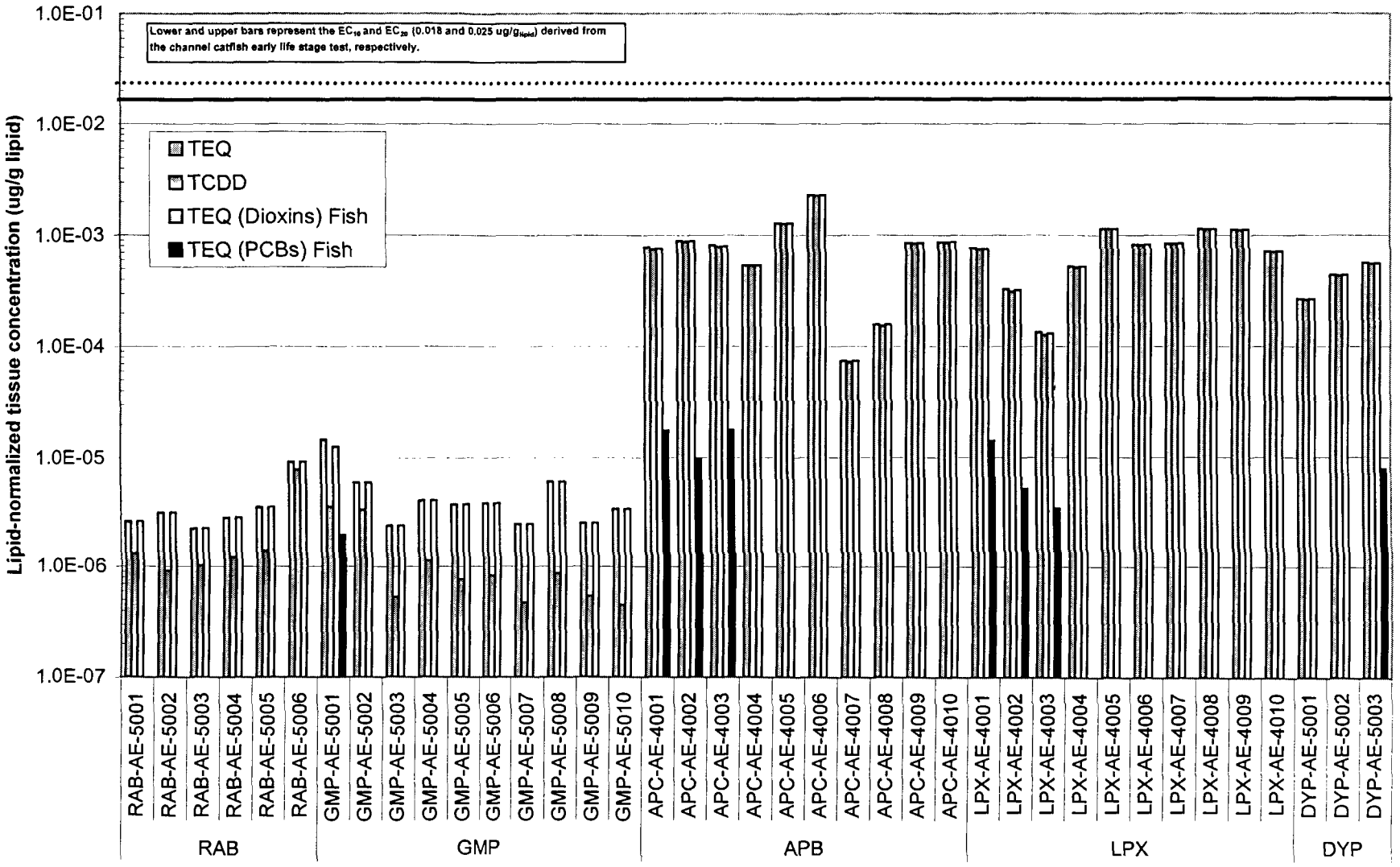
MACTEC Engineering and Consulting, Inc.
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9/20/2004

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FIGURE 76
COMPARISON OF TCDD AND TEQs IN AMERICAN EEL TISSUE TO ELS EFFECT THRESHOLD CONCENTRATIONS

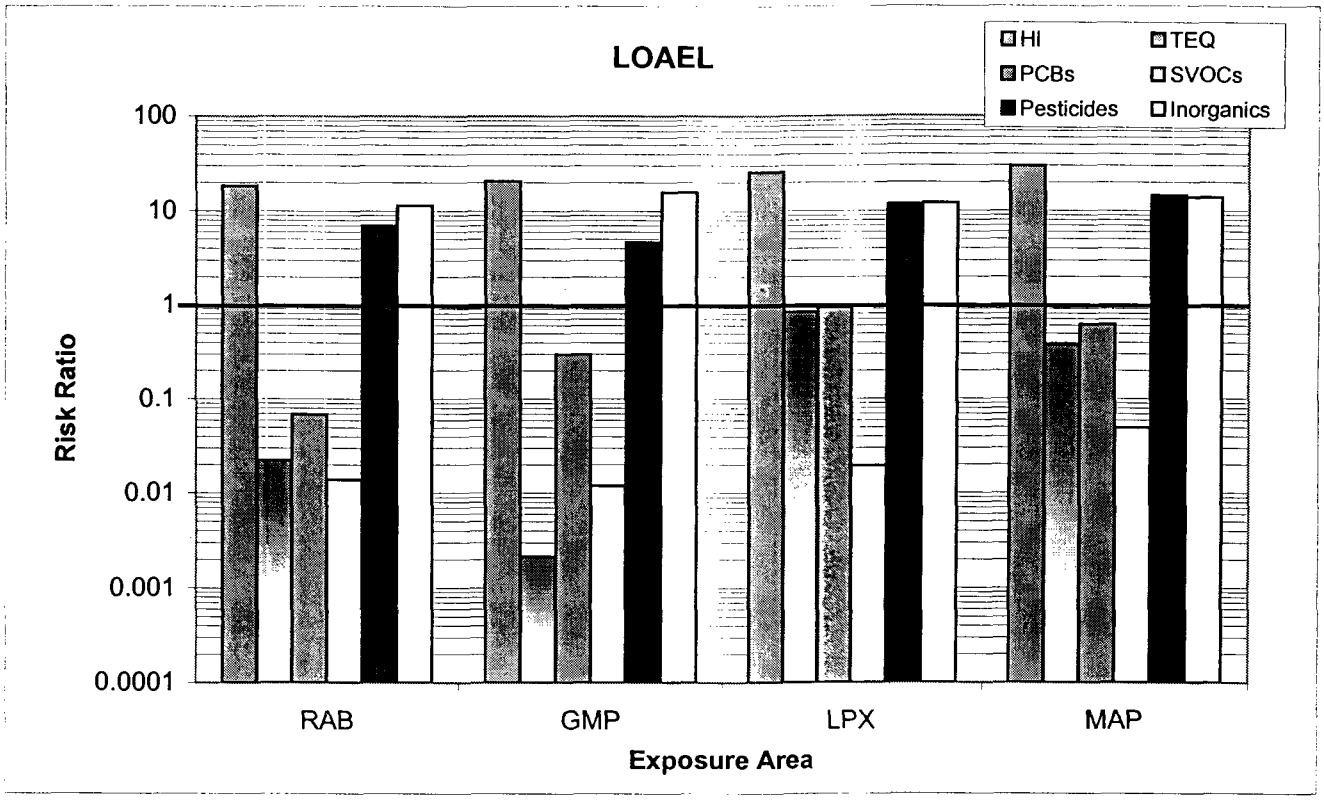
Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

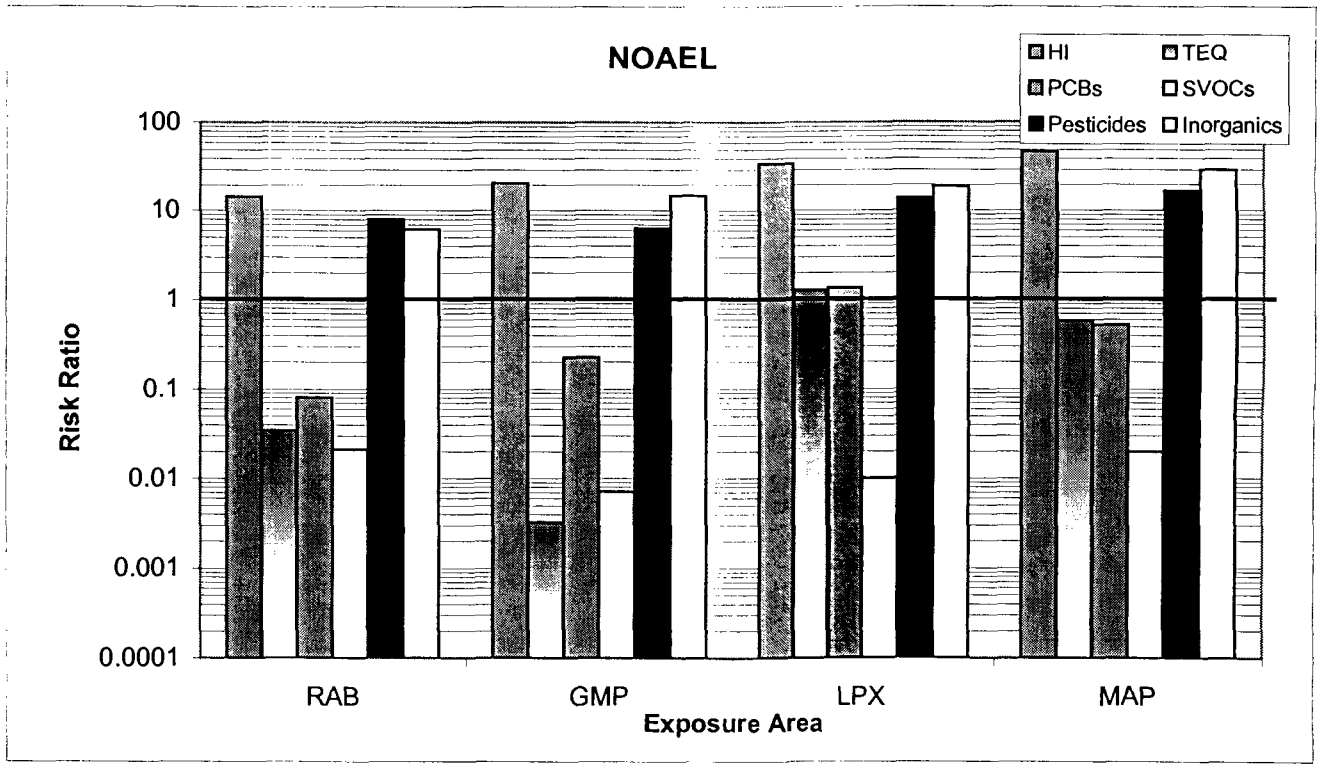


MACTEC Engineering and Consulting, Inc.
51226.24
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NormalizedFish-Egg(JUL04), Eel

FIGURE 77
CBR RISK RATIOS BY EXPOSURE AREA FOR LARGEMOUTH BASS SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

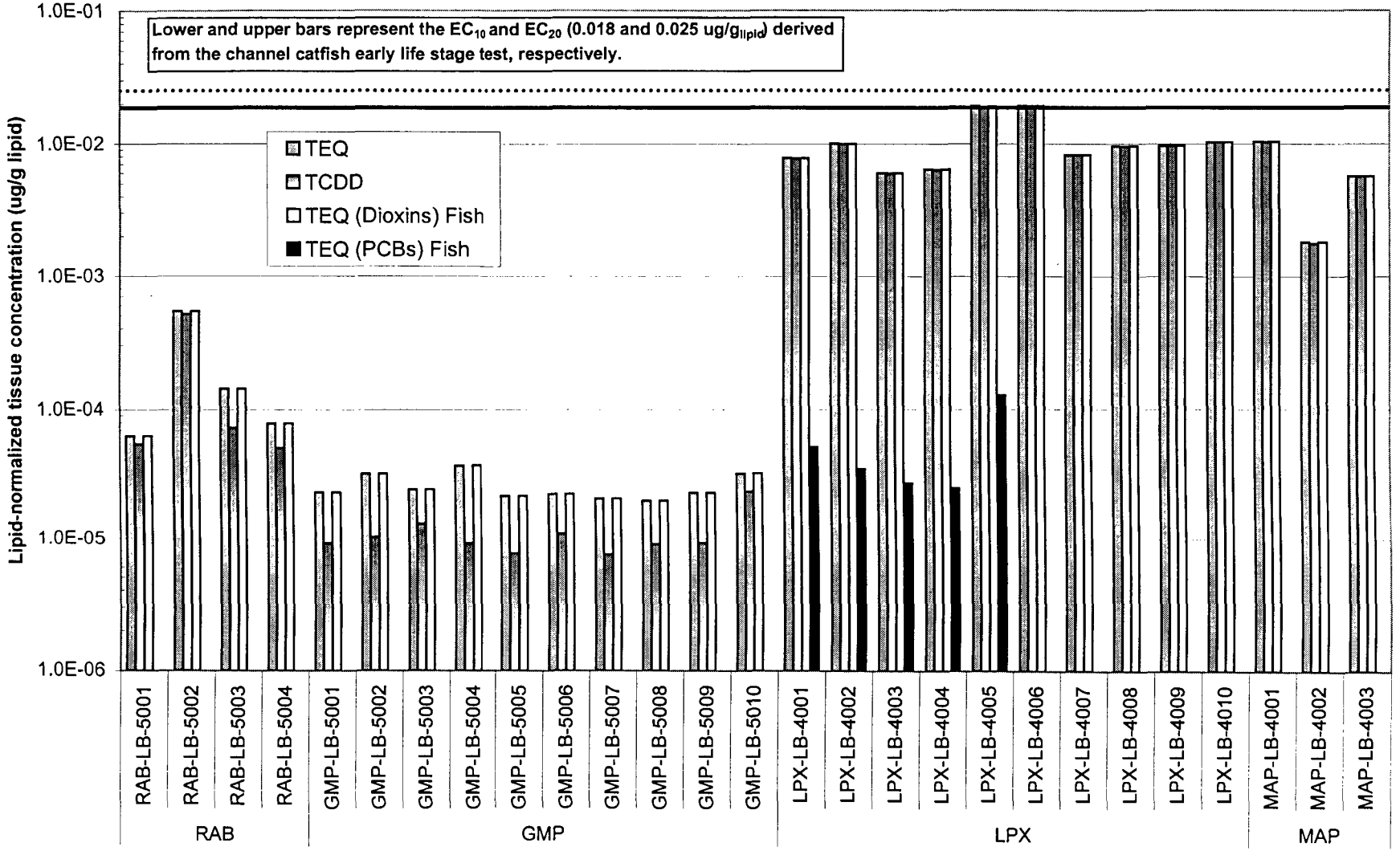




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FIGURE 78
COMPARISON OF TCDD AND TEQs IN LARGEMOUTH BASS TISSUE TO ELS EFFECT THRESHOLD CONCENTRATIONS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

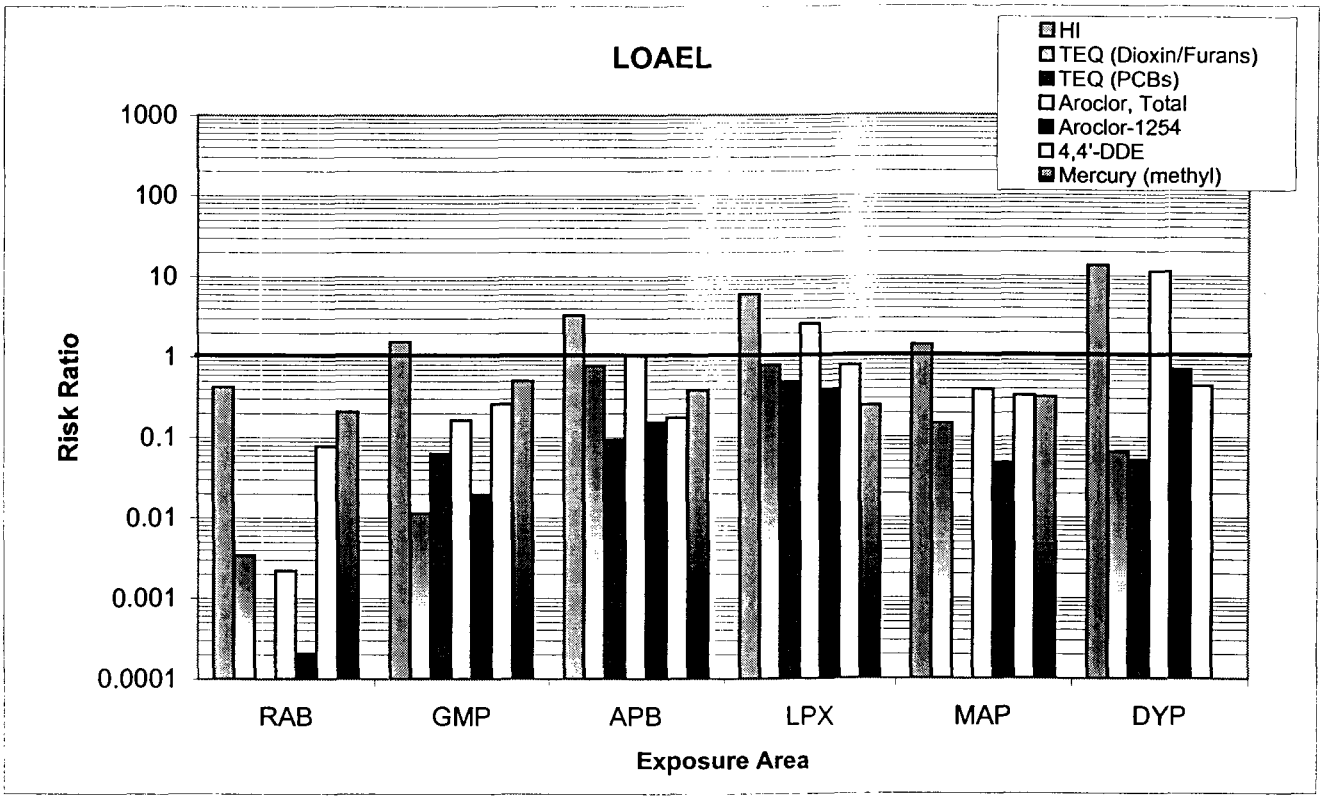


MACTEC Engineering and Consulting, Inc.
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NormalizedFish-Egg(JUL04), Bass

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FIGURE 79
SUMMARY OF WILDLIFE EXPOSURE MODELING - GREAT BLUE HERON

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



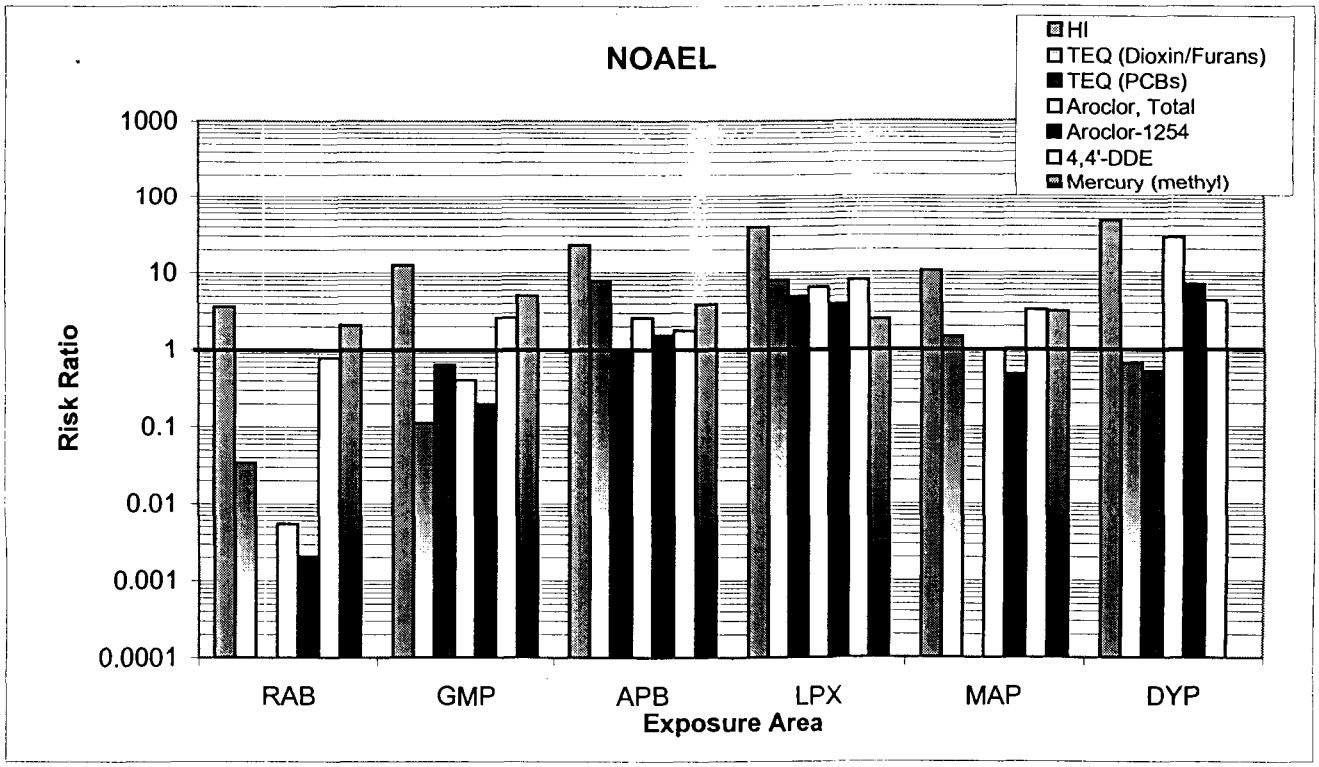
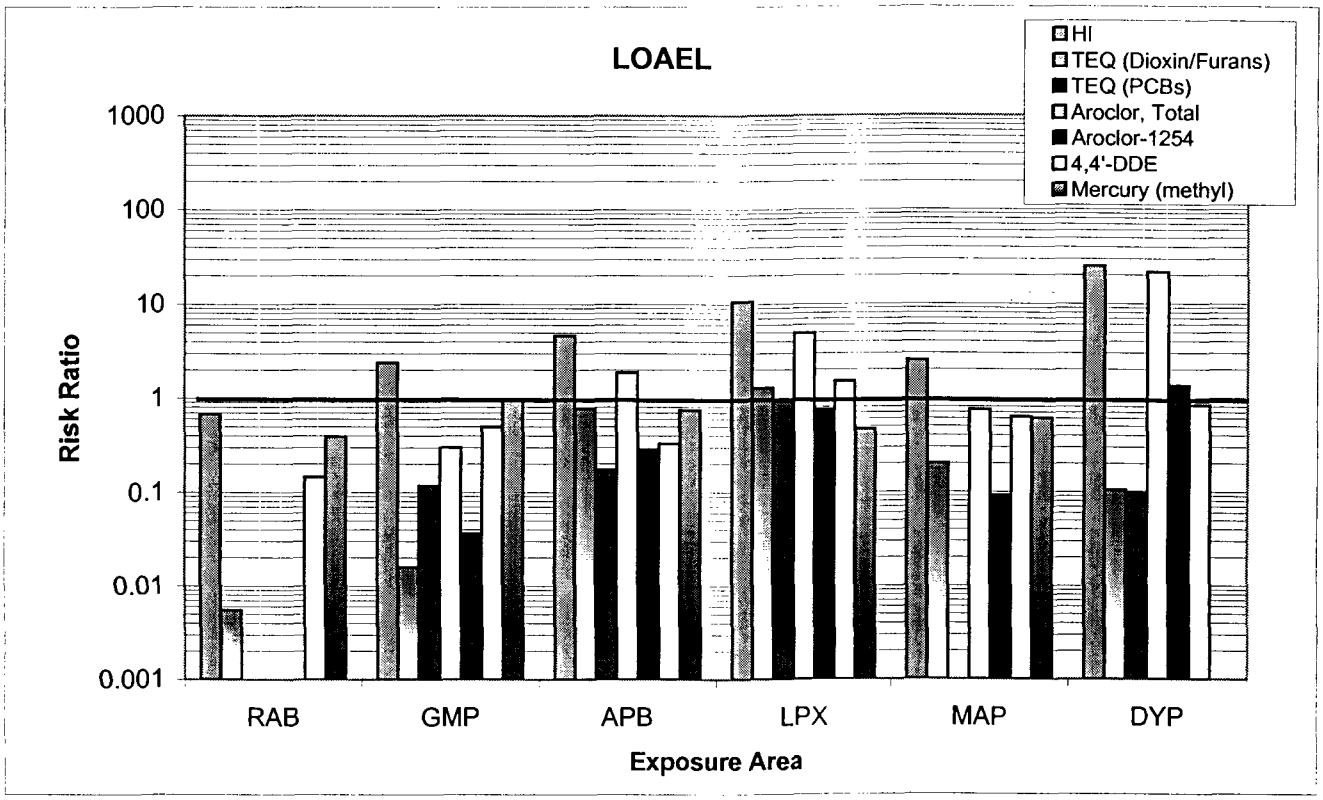
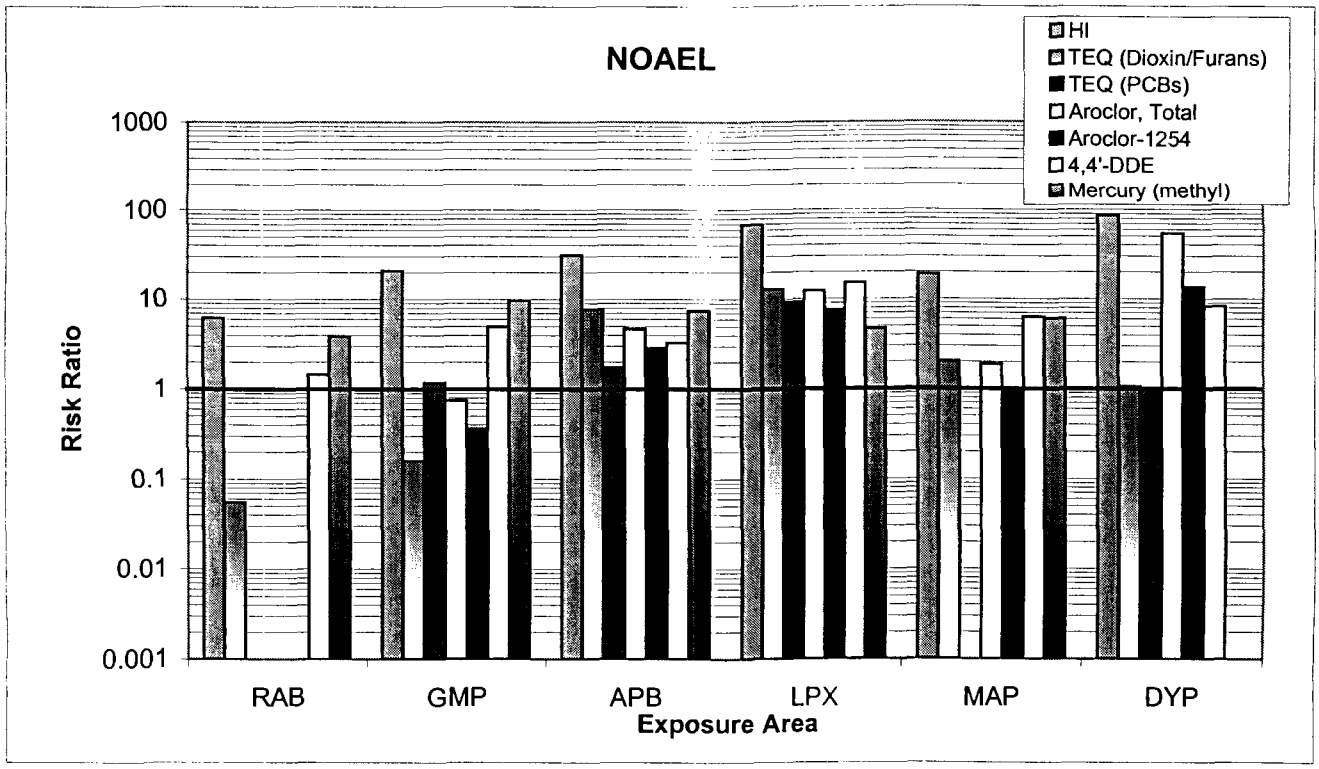


FIGURE 80
SUMMARY OF WILDLIFE EXPOSURE MODELING - KINGFISHER

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

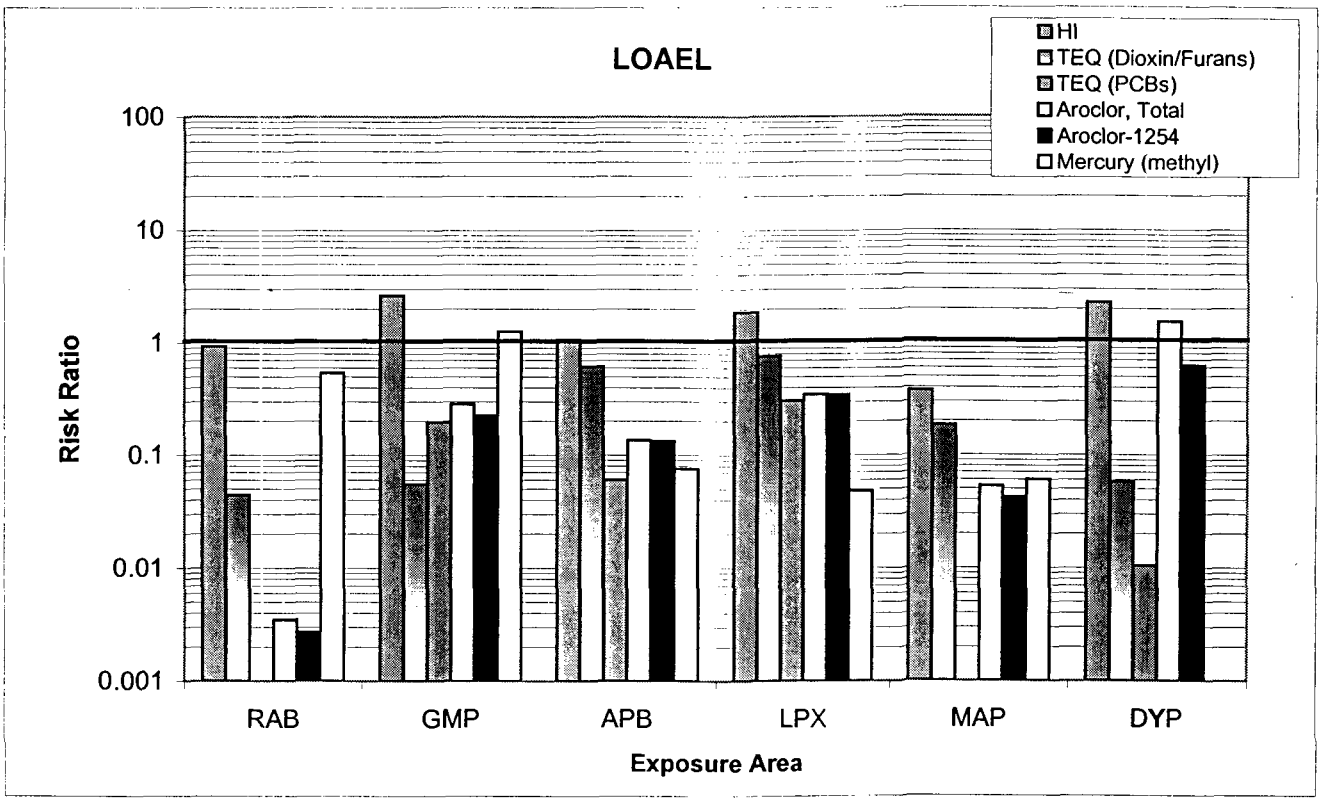


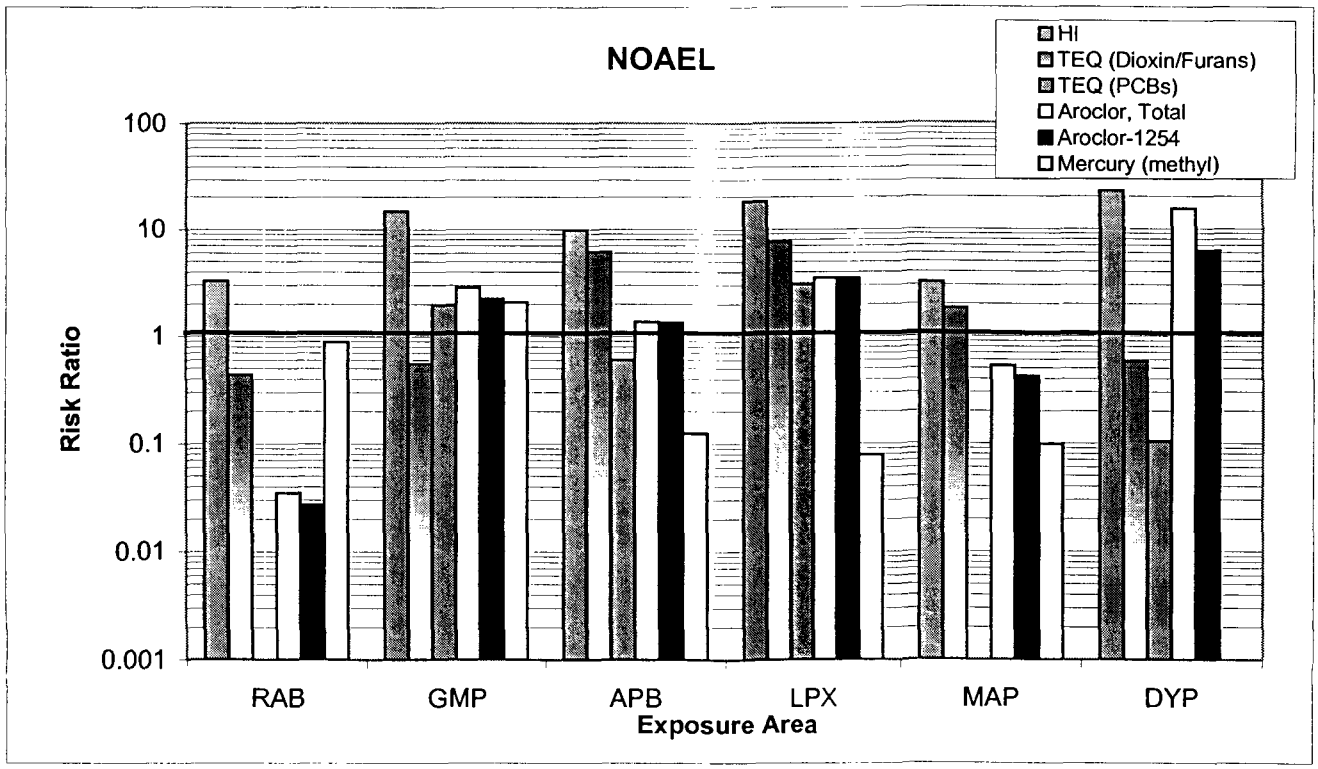


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FIGURE 81
SUMMARY OF WILDLIFE EXPOSURE MODELING - RIVER OTTER

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



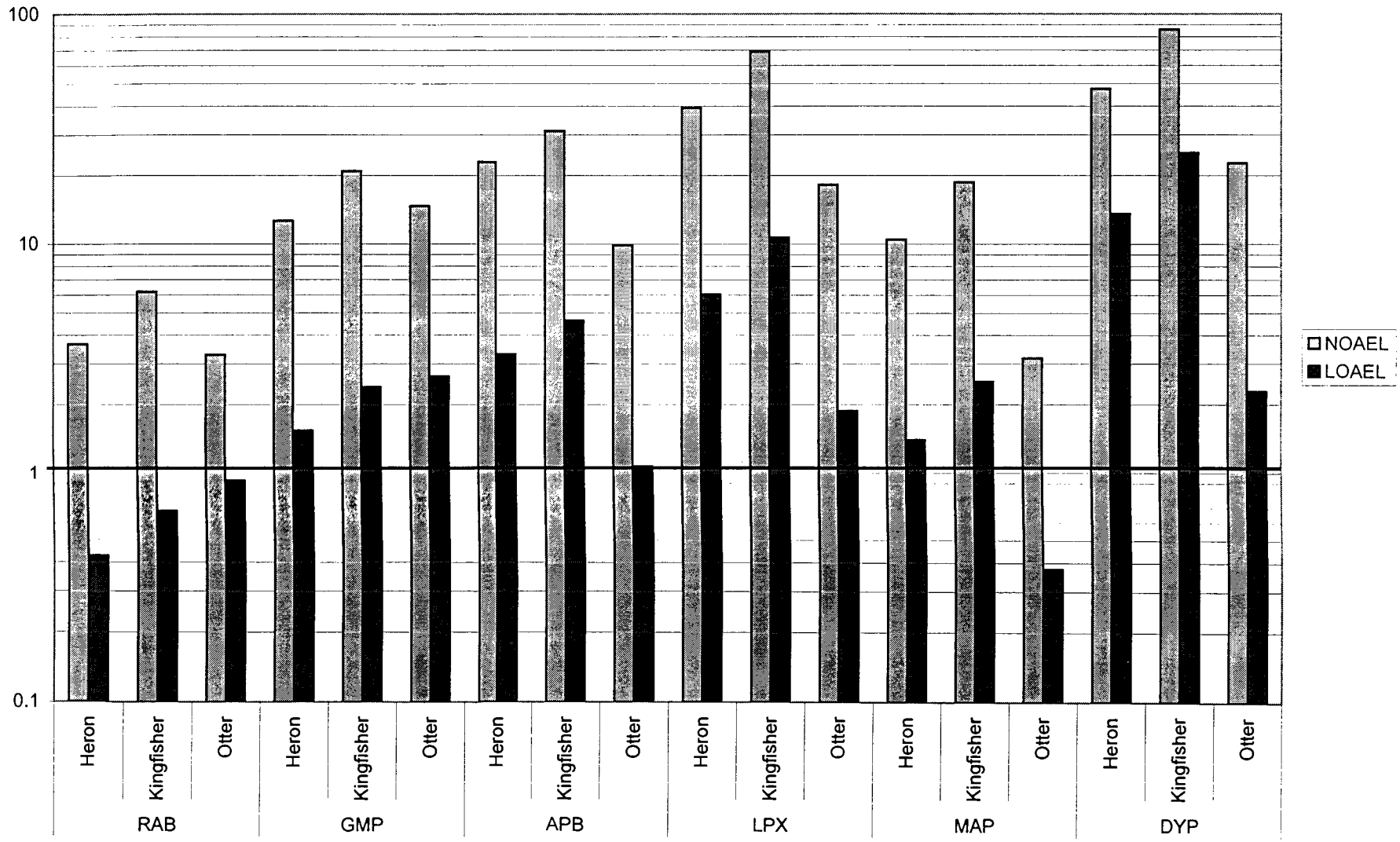


US EPA ARCHIVE DOCUMENT

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FIGURE 82
FOOD WEB MODELING SUMMARY - PISCIVOROUS WILDLIFE RECEPTORS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

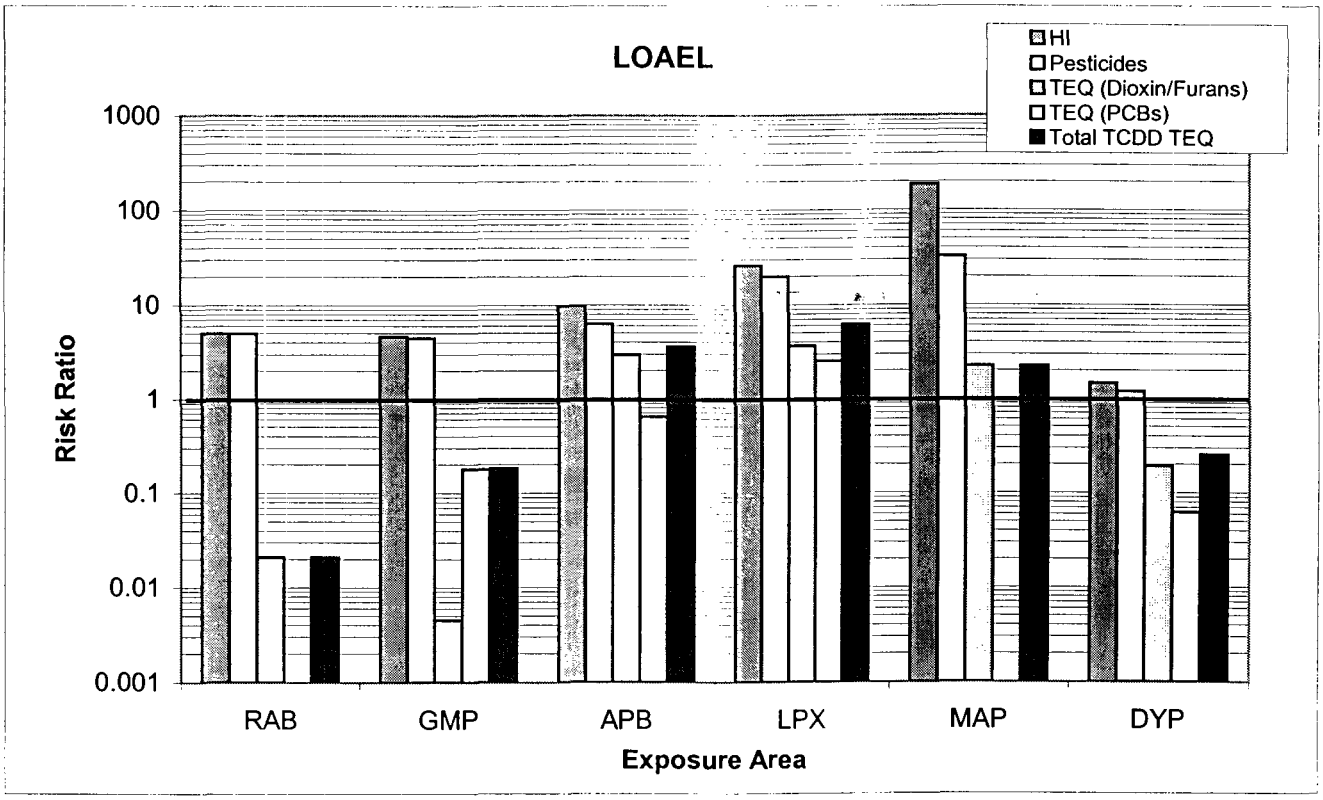


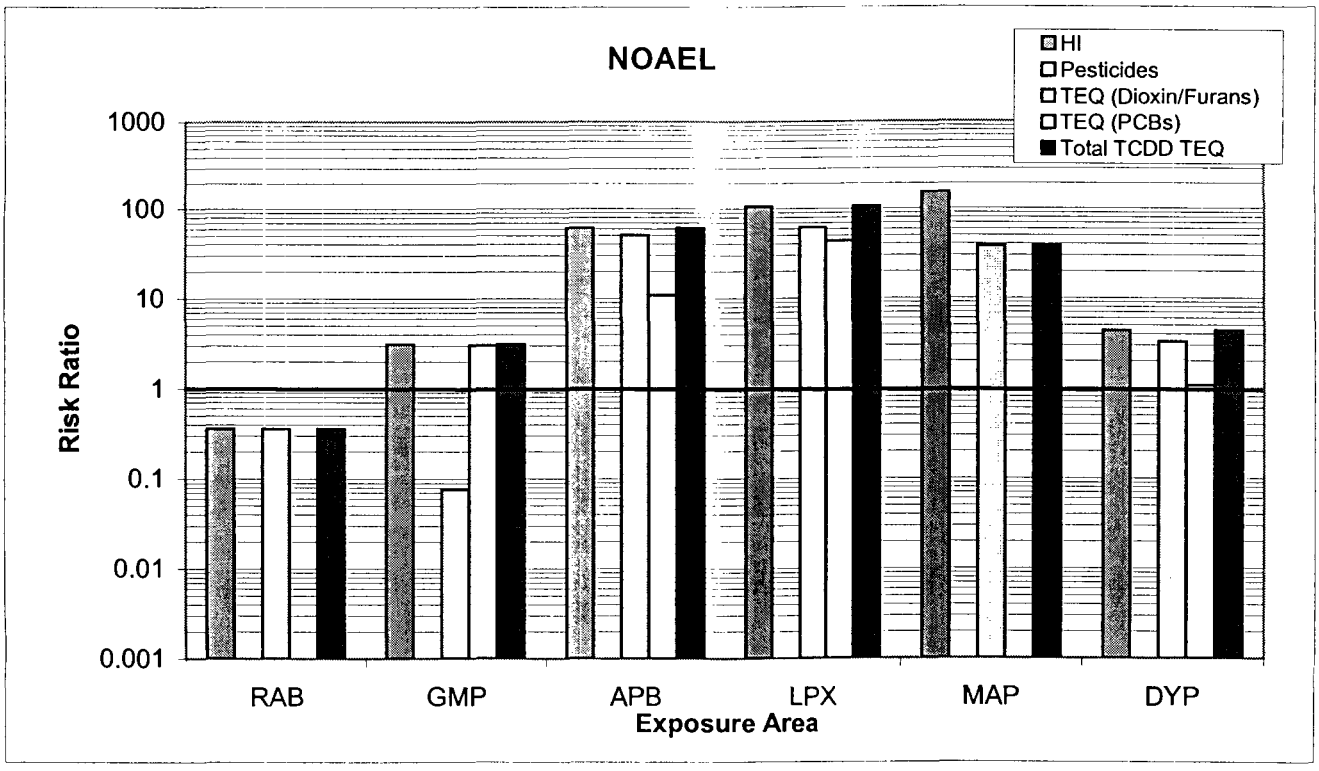
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Piscivore Risk Summary(JUL04)Piscivore Risk Summary

8/5/2004

FIGURE 83
CBR RISK RATIOS - ESTIMATED PISCIVOROUS BIRD EGG TISSUE

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

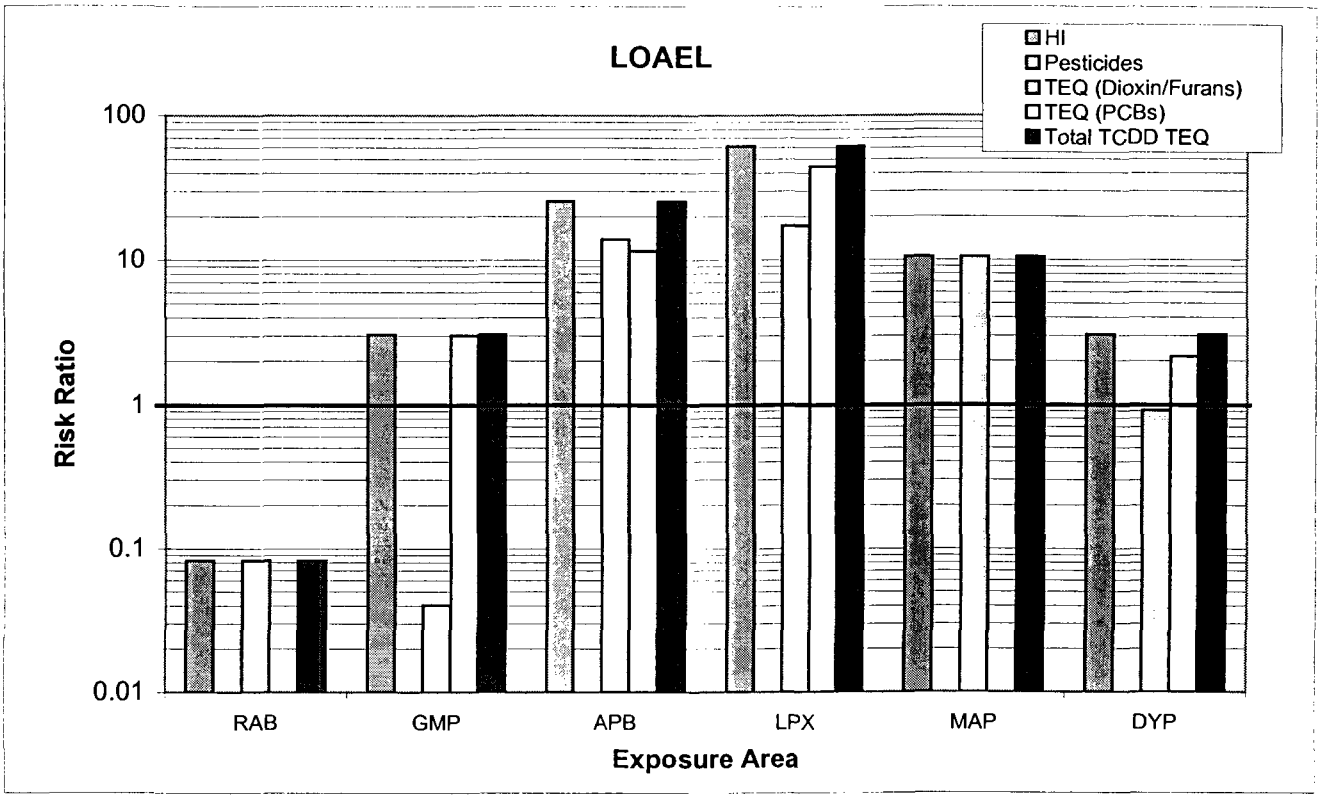


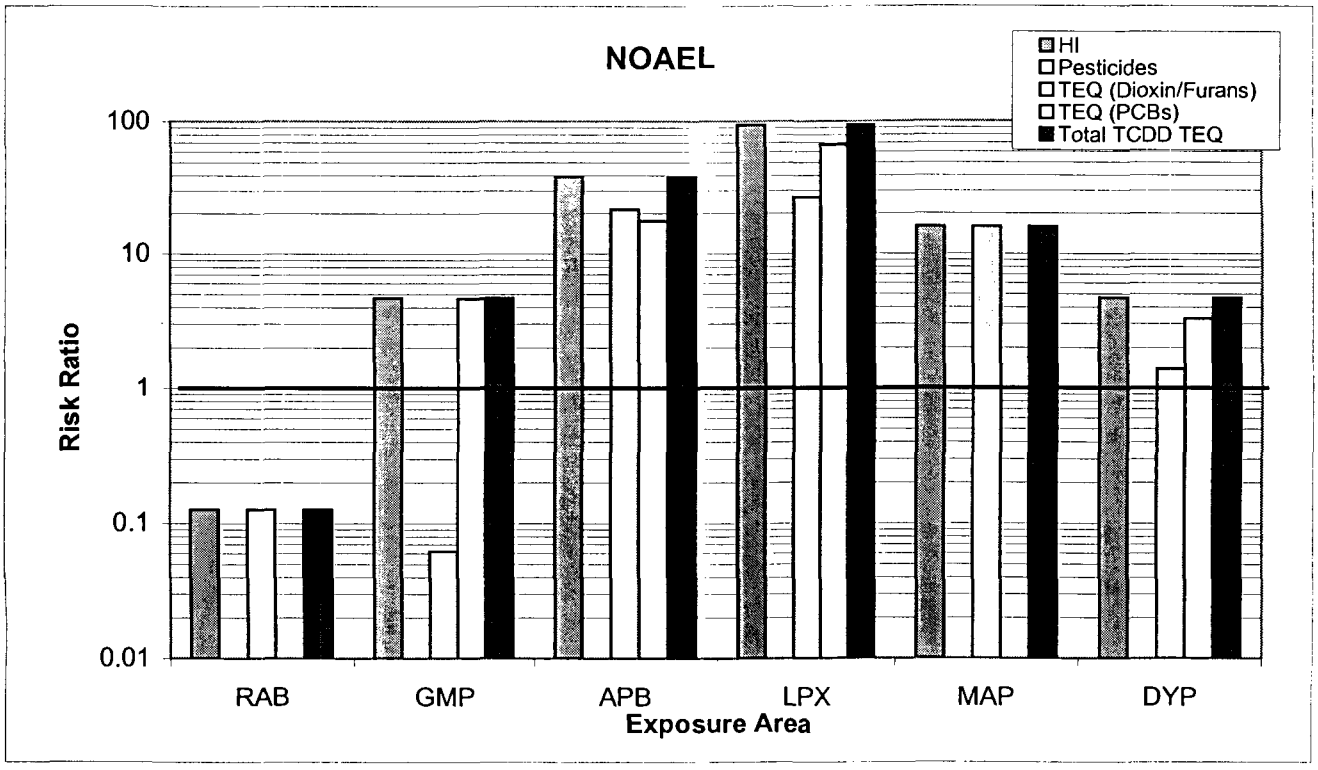


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FIGURE 84
CBR RISK RATIOS - ESTIMATED PISCIVOROUS MAMMAL TISSUE

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



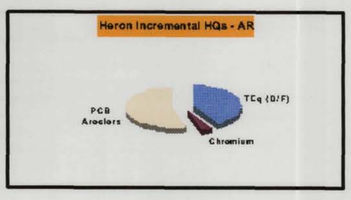


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Fish-BMF-TEFs(rev)Mammal

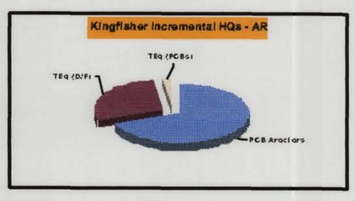
8/5/2004

US EPA ARCHIVE DOCUMENT

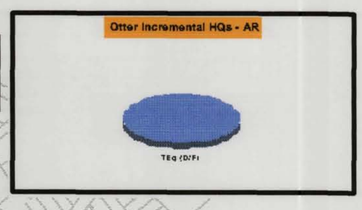
Incremental HQs - Heron (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	7.6E+00	7.6E-01	42%
Chromium	3.7E-01	7.5E-02	4%
PCB Aroclors	3.5E+00	1.0E+00	54%
Incremental HI	**1.2E+01**	**1.8E+00**	



Incremental HQs - Kingfisher (AR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	6.4E+00	1.8E+00	69%
TEq (D/F)	7.6E+00	7.6E-01	29%
TEq (PCBs)	5.5E-01	5.5E-02	2%
Incremental HI	**1.5E+01**	**2.6E+00**	



Incremental HQs - Otter (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	5.6E+00	5.6E-01	100%
Incremental HI	**5.6E+00**	**5.6E-01**	

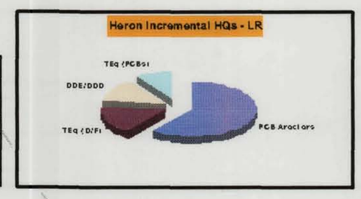


ALLENDALE POND

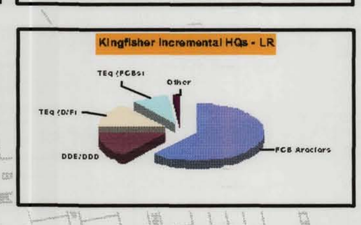
ALLENDALE REACH

ALLENDALE DAM

Incremental HQs - Heron (LR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	9.7E+00	2.8E+00	60%
TEq (D/F)	7.8E+00	7.8E-01	17%
DDE/DDD	6.6E+00	6.6E-01	14%
TEq (PCBs)	4.3E+00	4.3E-01	9%
Incremental HI	**2.8E+01**	**4.6E+00**	



Incremental HQs - Kingfisher (LR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	1.9E+01	5.3E+00	61%
DDE/DDD	1.2E+01	1.2E+00	14%
TEq (D/F)	1.3E+01	1.3E+00	14%
TEq (PCBs)	8.1E+00	8.1E-01	9%
Other	1.2E+00	1.2E-01	1%
Incremental HI	**5.3E+01**	**8.8E+00**	



LYMAN MILL REACH

ASSAPUMPSET POND

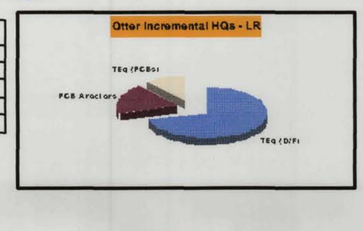
ASSAPUMPSET BROOK

WOONASQUATUCKET RIVER

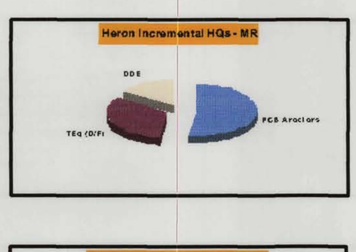
LYMAN MILL POND

LYMAN MILL DAM

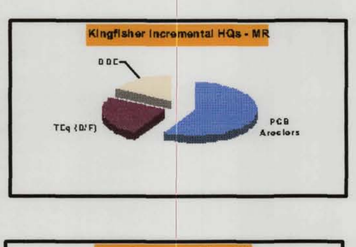
Incremental HQs - Otter (LR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	7.0E+00	7.0E-01	71%
PCB Aroclors	1.8E+00	1.8E-01	18%
TEq (PCBs)	1.1E+00	1.1E-01	11%
Incremental HI	**1.0E+01**	**1.0E+00**	



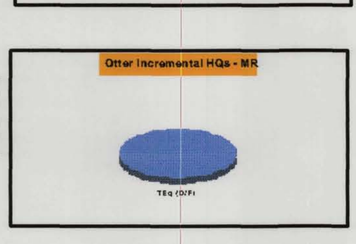
Incremental HQs - Heron (MR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	5.6E-01	2.2E-01	52%
TEq (D/F)	1.4E+00	1.4E-01	32%
DDE	6.6E-01	6.6E-02	15%
Incremental HI	**2.6E+00**	**4.3E-01**	



Incremental HQs - Kingfisher (MR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	1.1E+00	4.4E-01	58%
TEq (D/F)	1.9E+00	1.9E-01	24%
DDE	1.3E+00	1.3E-01	17%
Incremental HI	**4.3E+00**	**7.6E-01**	



Incremental HQs - Otter (MR)			
Analyte	RME	CT	%
TEq (D/F)	1.3E+00	1.3E-01	100%
Incremental HI	**1.3E+00**	**1.3E-01**	

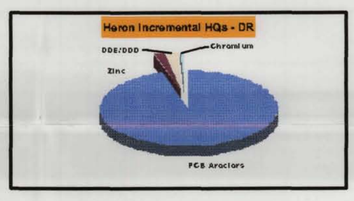


MANTON REACH

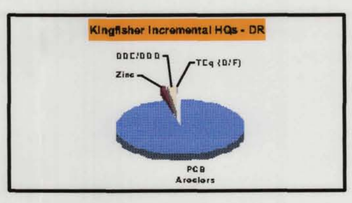
MANTON POND

MANTON DAM

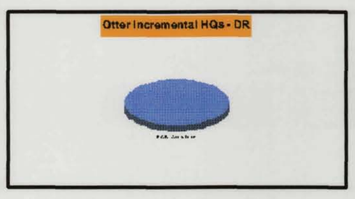
Incremental HQs - Heron (DR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	3.5E+01	1.2E+01	95%
Zinc	2.0E+00	2.2E-01	2%
DDE/DDD	2.6E+00	2.6E-01	2%
Chromium	6.2E-01	1.2E-01	1%
Incremental HI	**4.0E+01**	**1.2E+01**	



Incremental HQs - Kingfisher (DR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	6.6E+01	2.3E+01	96%
Zinc	3.1E+00	3.4E-01	1%
DDE/DDD	4.9E+00	4.9E-01	2%
TEq (D/F)	9.0E-01	9.0E-02	0%
Incremental HI	**7.5E+01**	**2.4E+01**	



Incremental HQs - Otter (DR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	1.7E+01	1.7E+00	100%
Incremental HI	**1.7E+01**	**1.7E+00**	



DYERVILLE REACH

DYERVILLE POND

DYERVILLE DAM

300 0 300 600

SCALE IN FEET

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107 Audubon Road
Building II, #301
Wakefield, MA 01880
781-245-6606

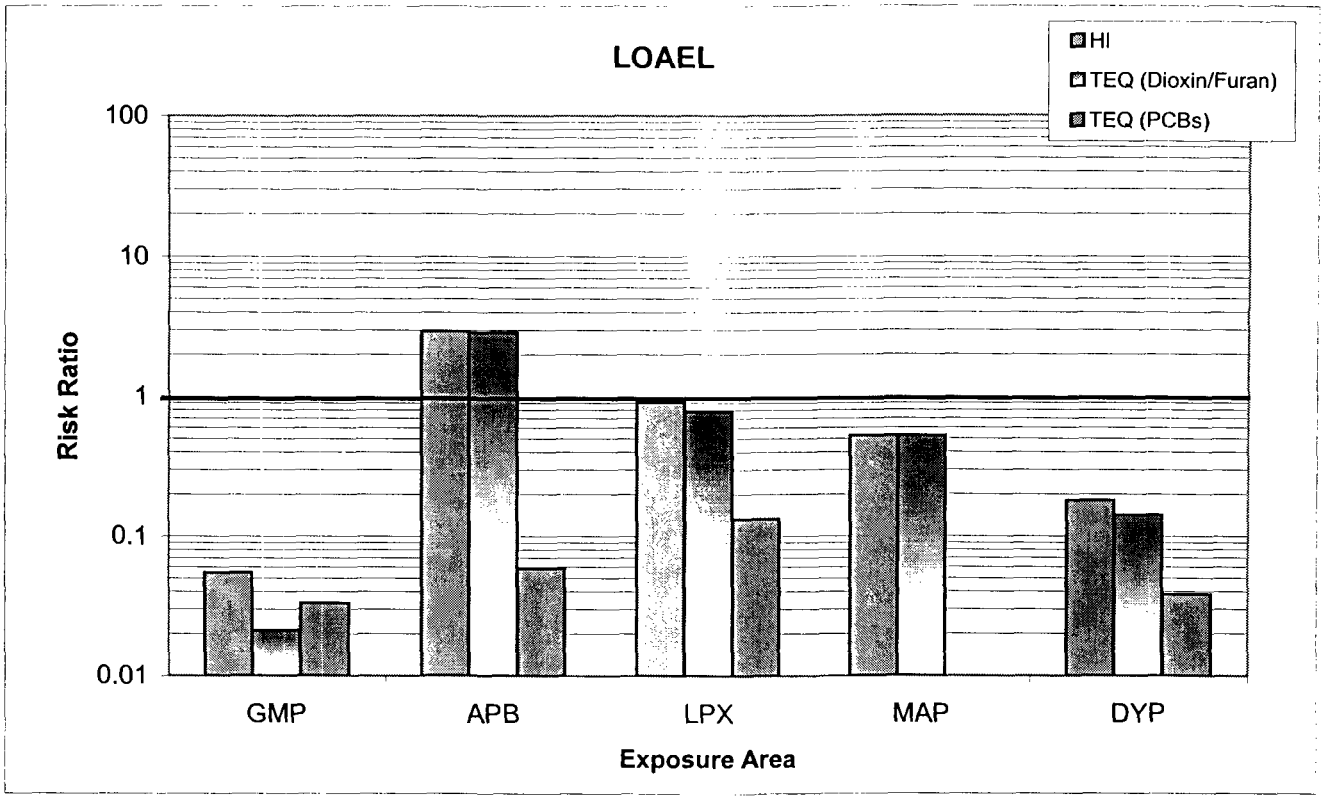
FIGURE 85
INCREMENTAL RISK SUMMARY FOR PISCIVOROUS WILDLIFE
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

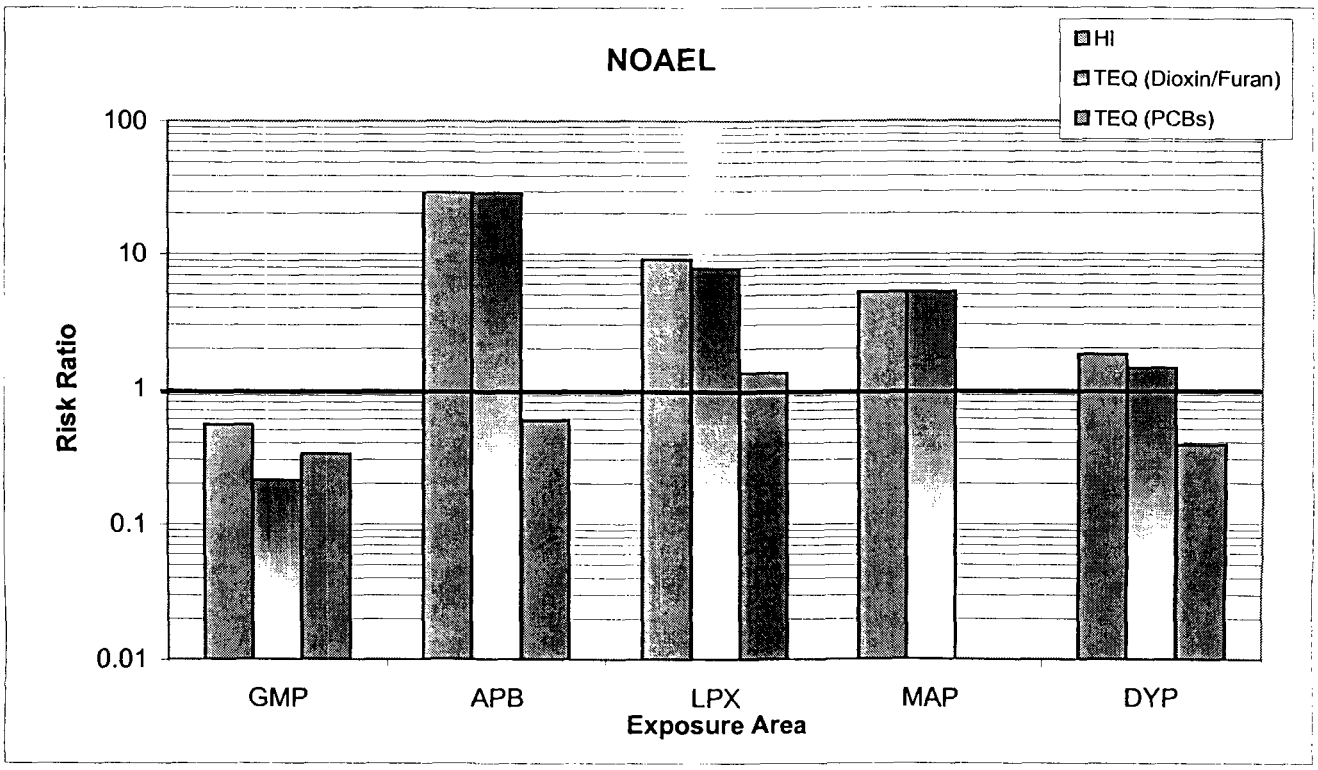
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US EPA ARCHIVE DOCUMENT

FIGURE 86
SUMMARY OF WILDLIFE EXPOSURE MODELING - TREE SWALLOW

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

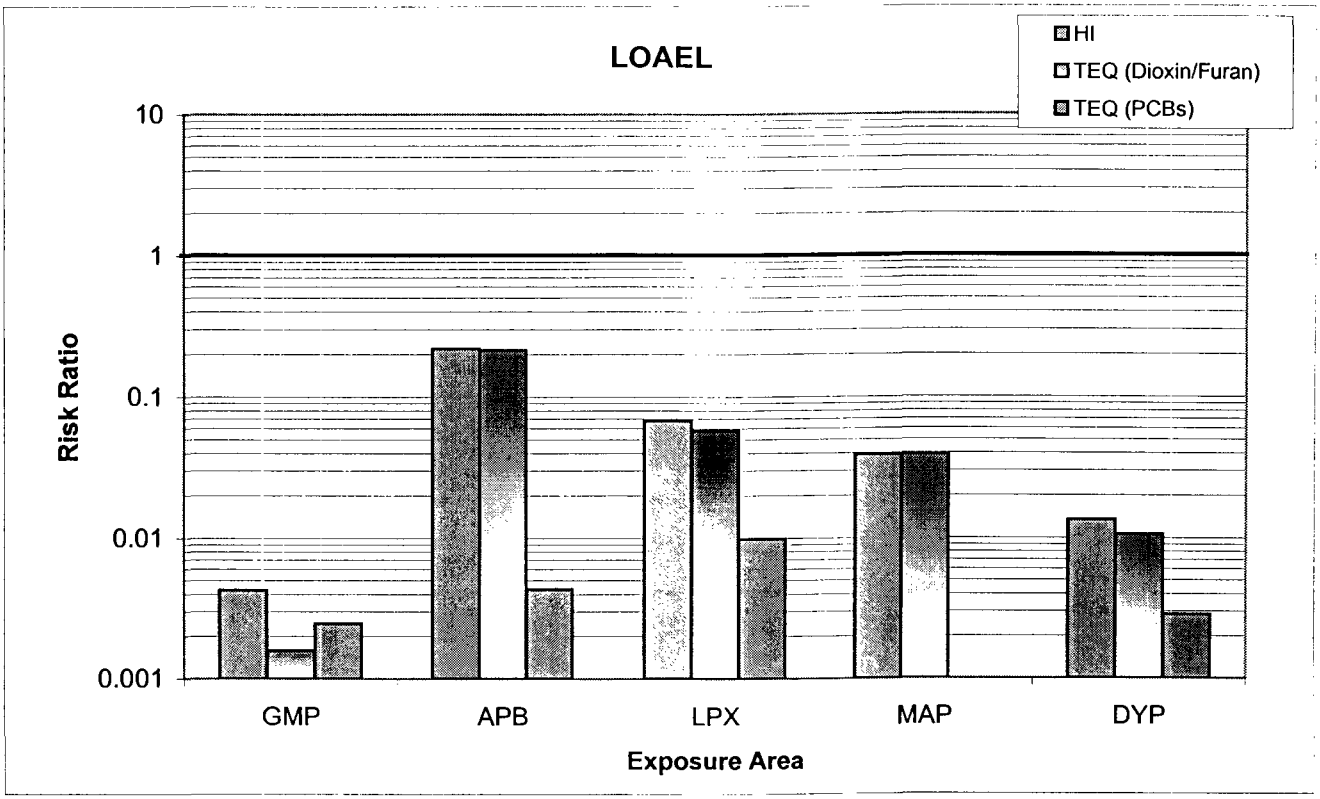


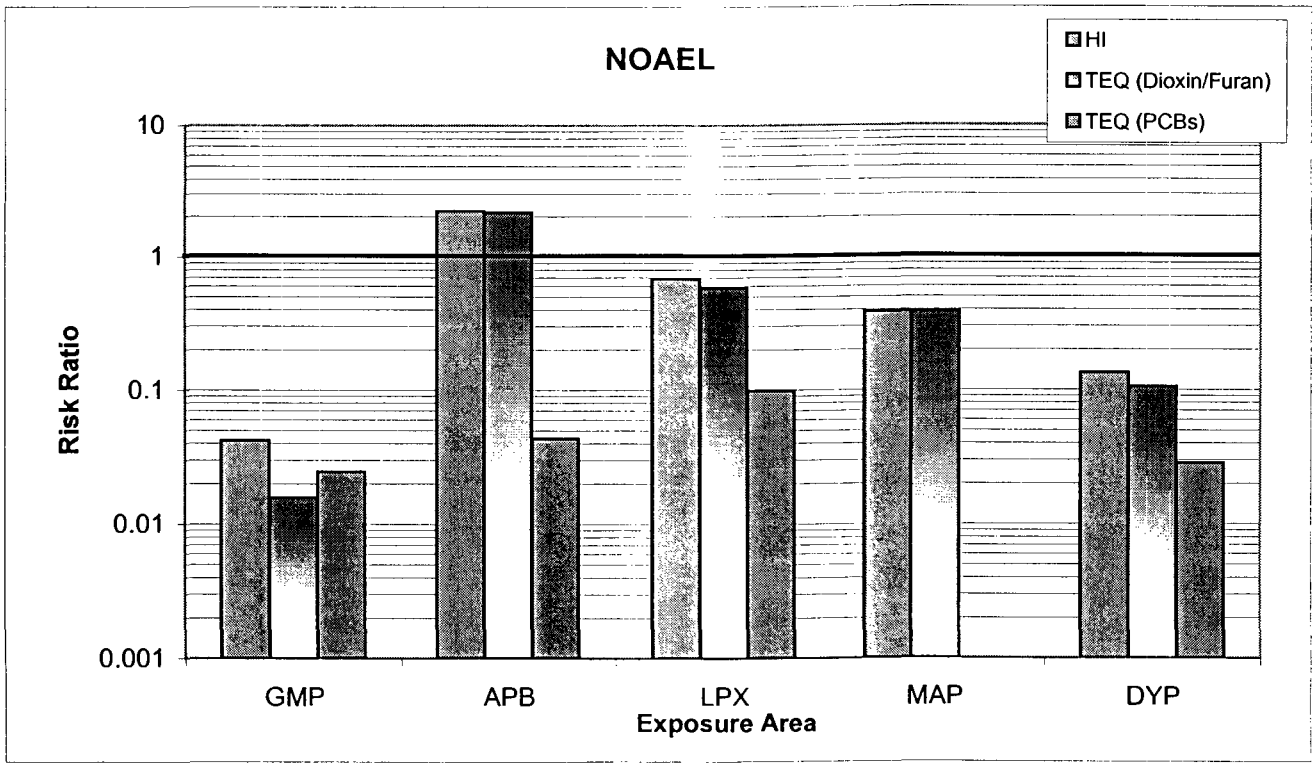


US EPA ARCHIVE DOCUMENT

FIGURE 87
SUMMARY OF WILDLIFE EXPOSURE MODELING - TREE SWALLOW NESTLING

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

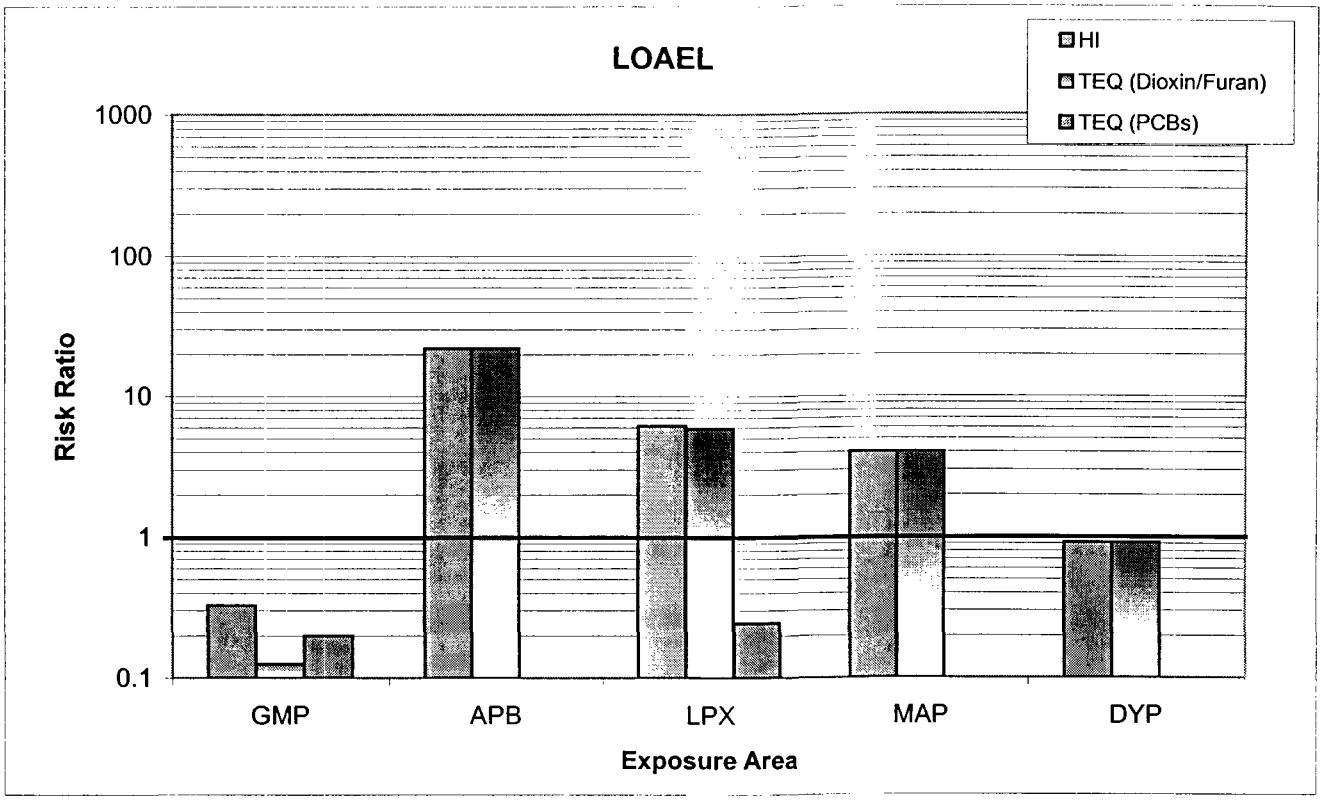


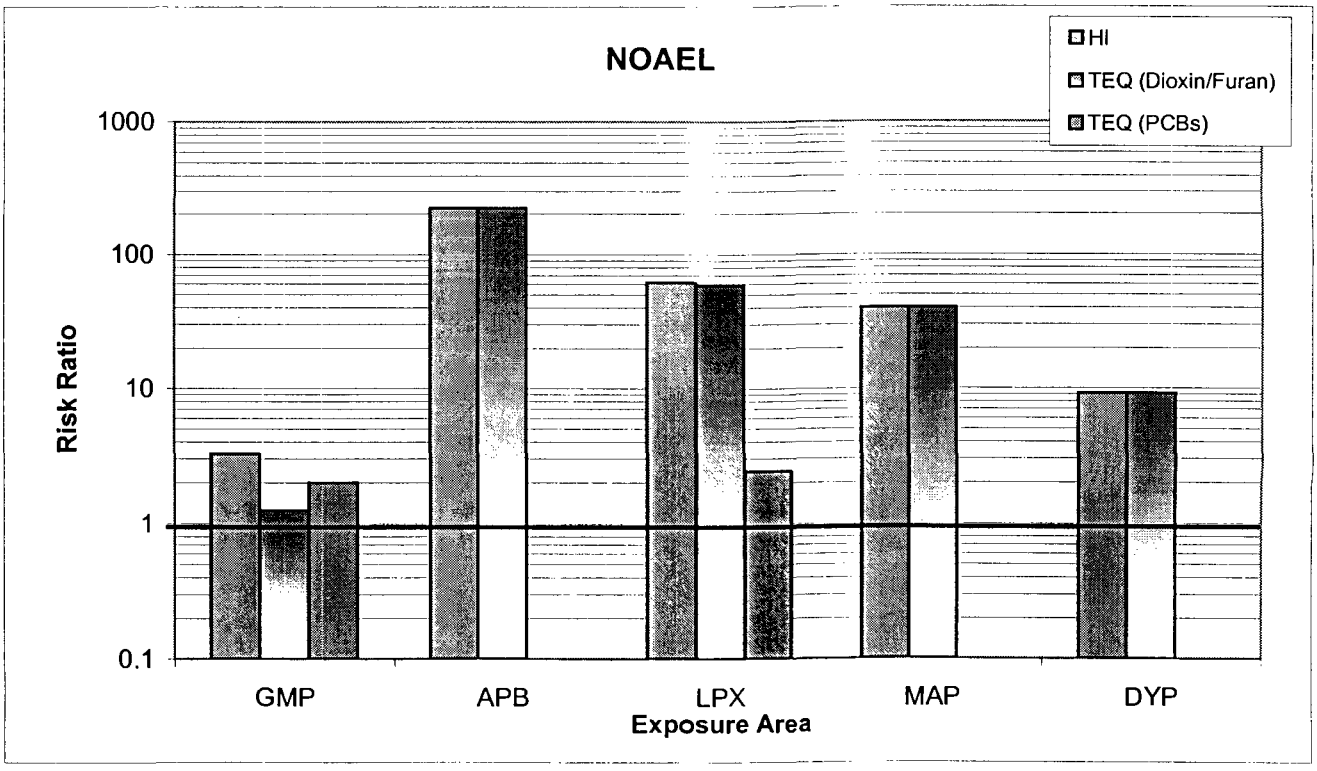


US EPA ARCHIVE DOCUMENT

FIGURE 88
SUMMARY OF WILDLIFE EXPOSURE MODELING - LITTLE BROWN BAT

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

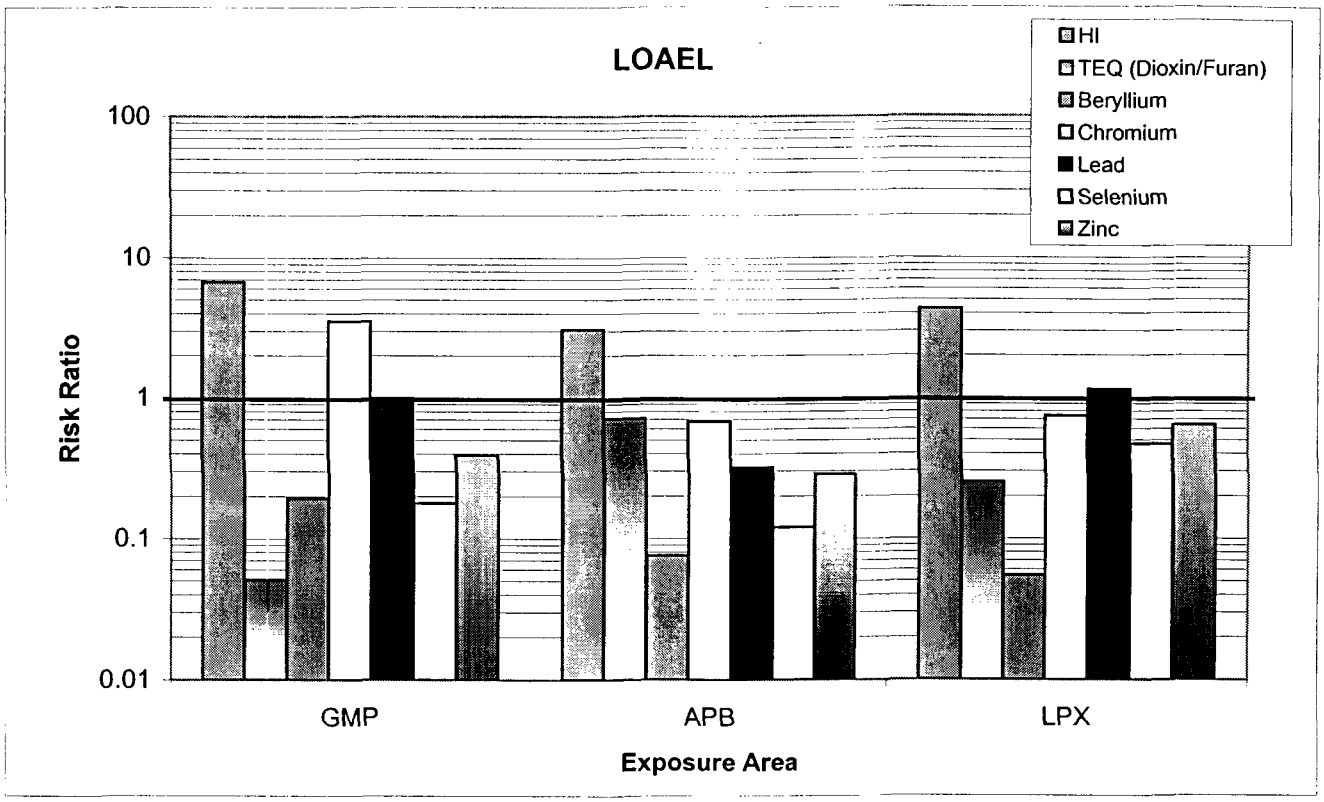


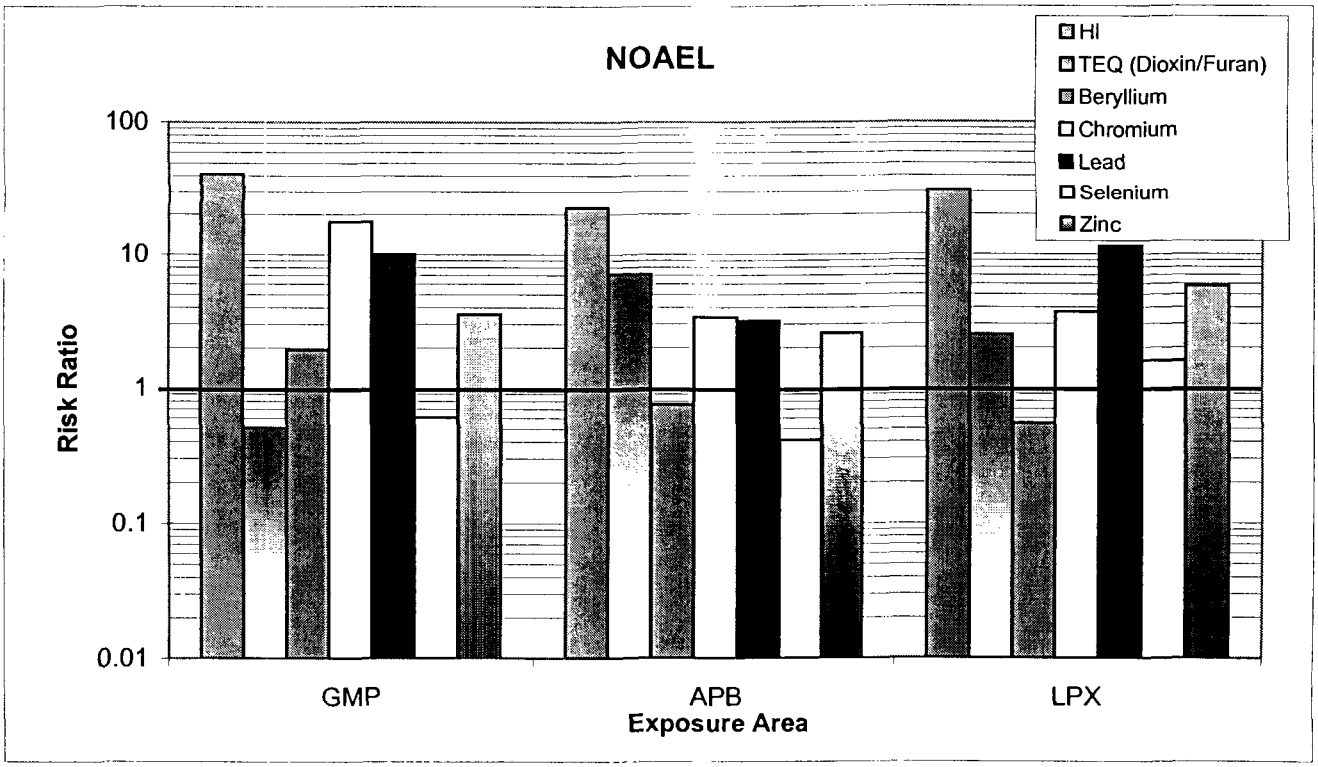


US EPA ARCHIVE DOCUMENT

FIGURE 89
SUMMARY OF WILDLIFE EXPOSURE MODELING - AMERICAN WOODCOCK

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





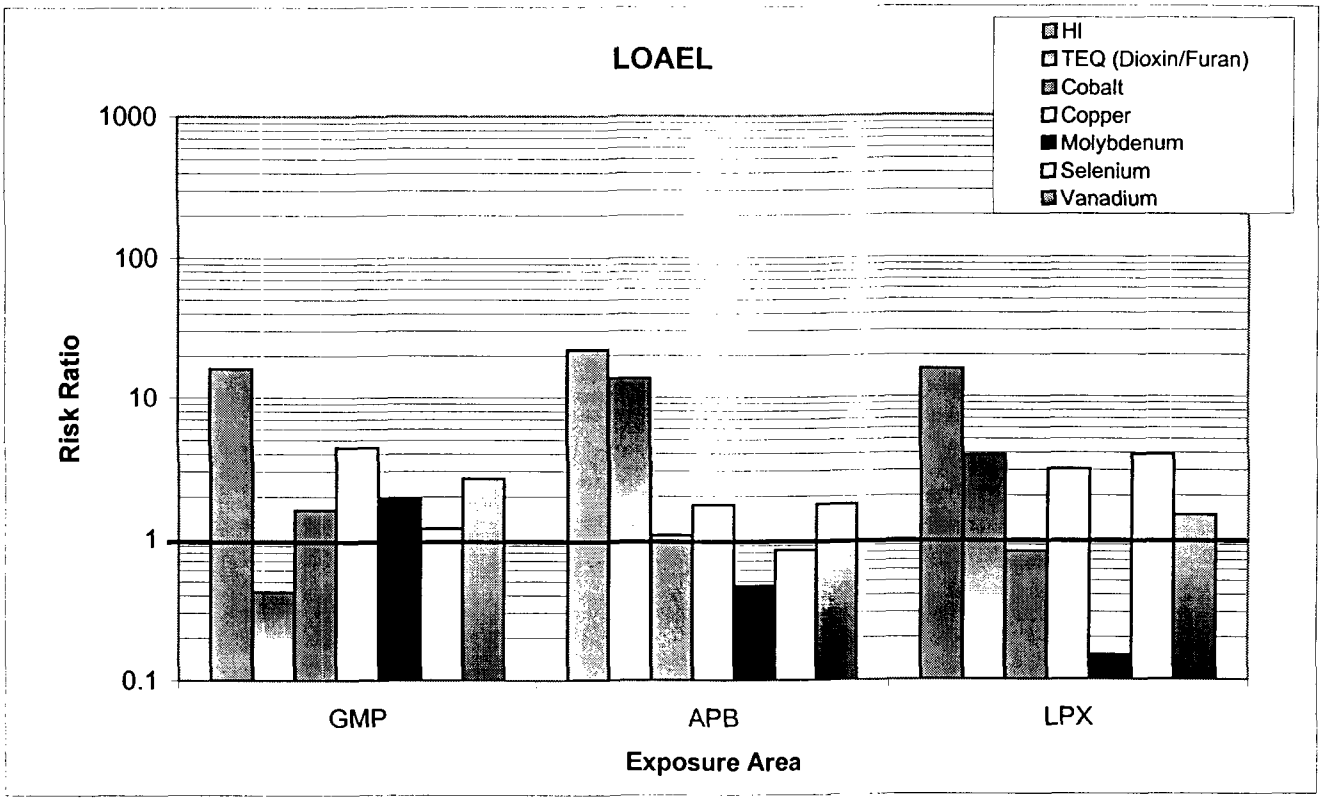
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51226.24
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Insectivore Risk Summary(JUL04)Woodcock

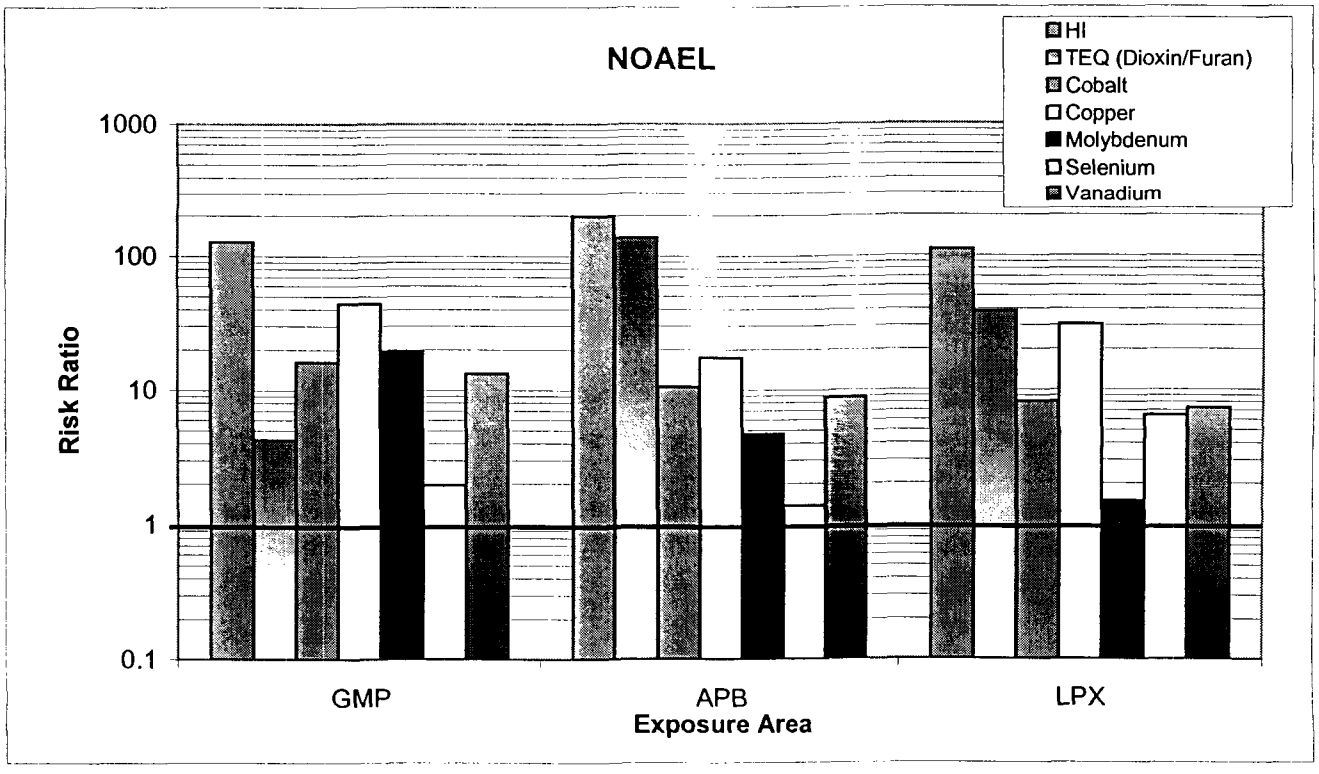
8/5/2004

US EPA ARCHIVE DOCUMENT

FIGURE 90
SUMMARY OF WILDLIFE EXPOSURE MODELING - SHORT-TAILED SHREW

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



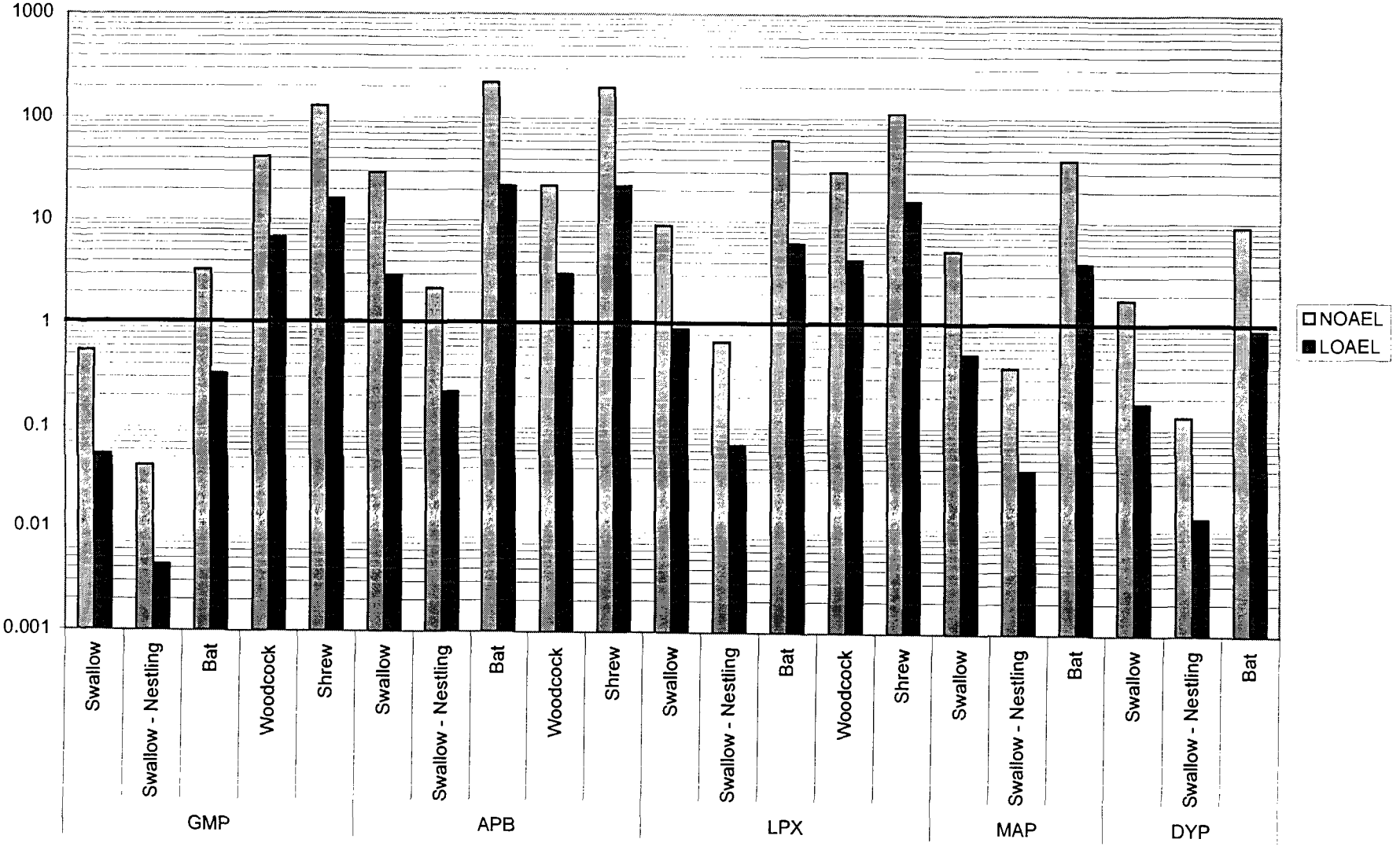


US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

FIGURE 91
FOOD WEB MODELING SUMMARY - INSECTIVOROUS/VERMIVOROUS WILDLIFE RECEPTORS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

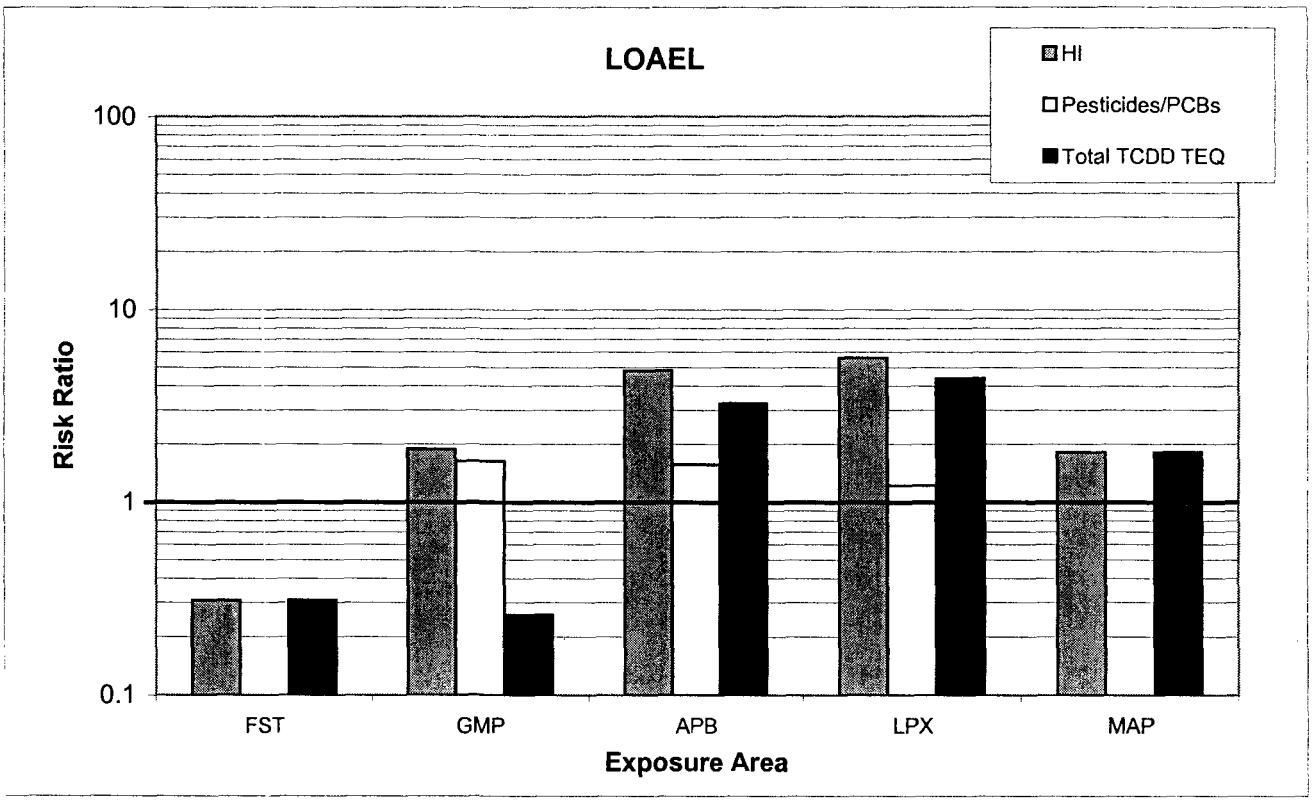


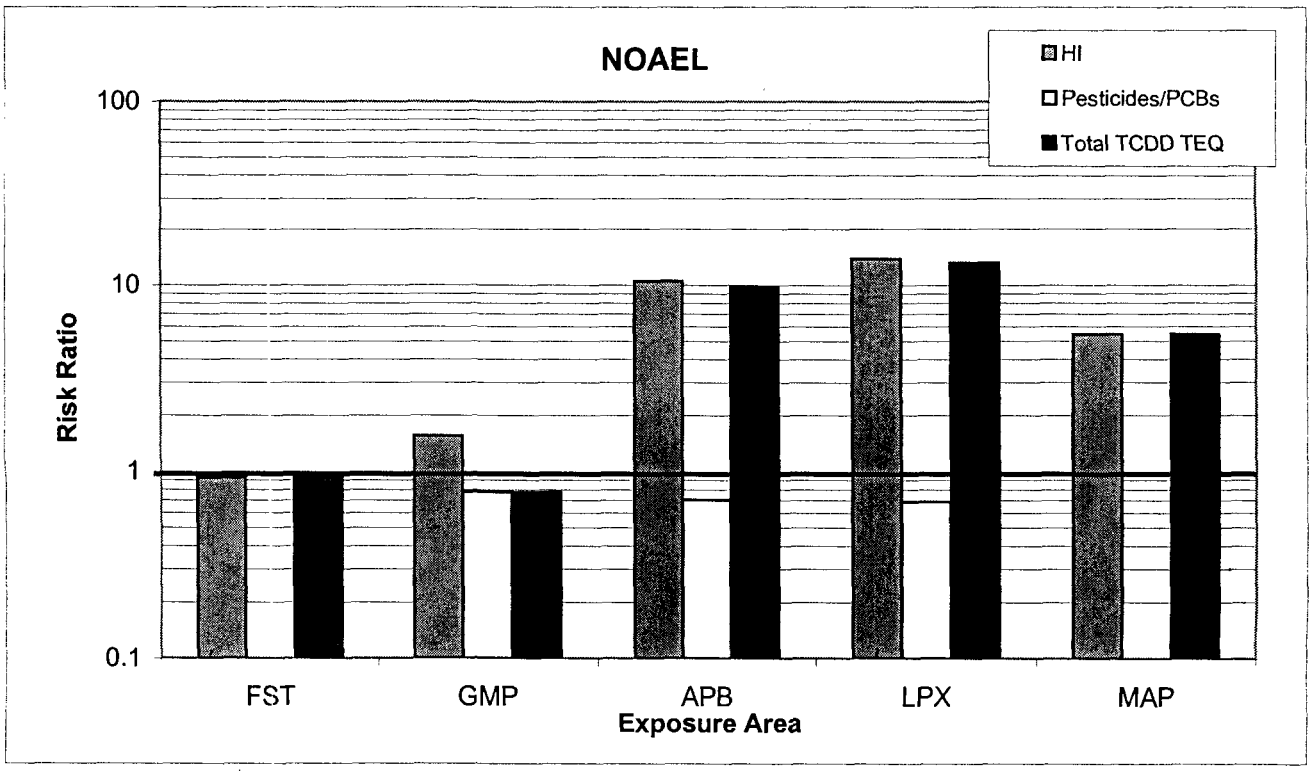
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Insectivore Risk Summary(JUL04)Insectivore Risk Summary

8/5/2004

FIGURE 92
CBR RISK RATIOS BY EXPOSURE AREA FOR SWALLOW EGG SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

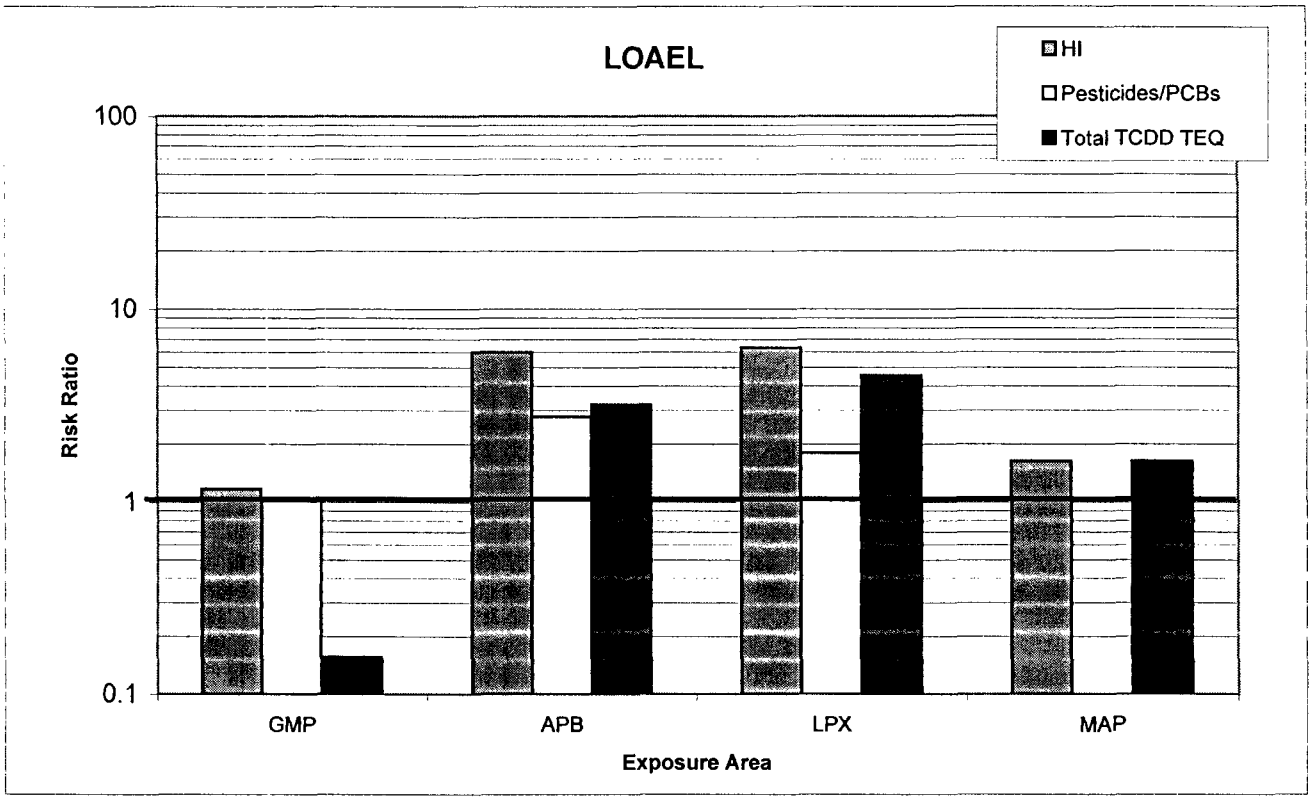




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FIGURE 93
CBR RISK RATIOS BY EXPOSURE AREA FOR SWALLOW NESTLING SAMPLES

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



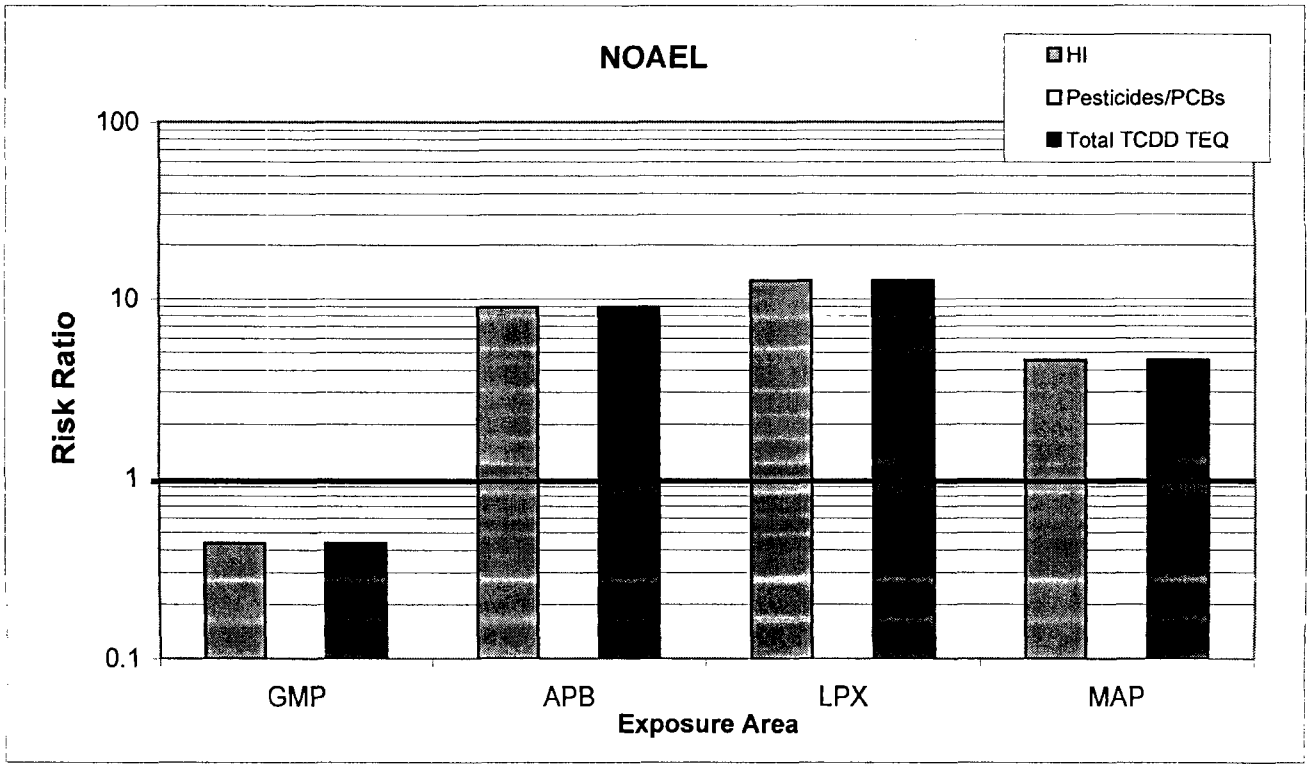
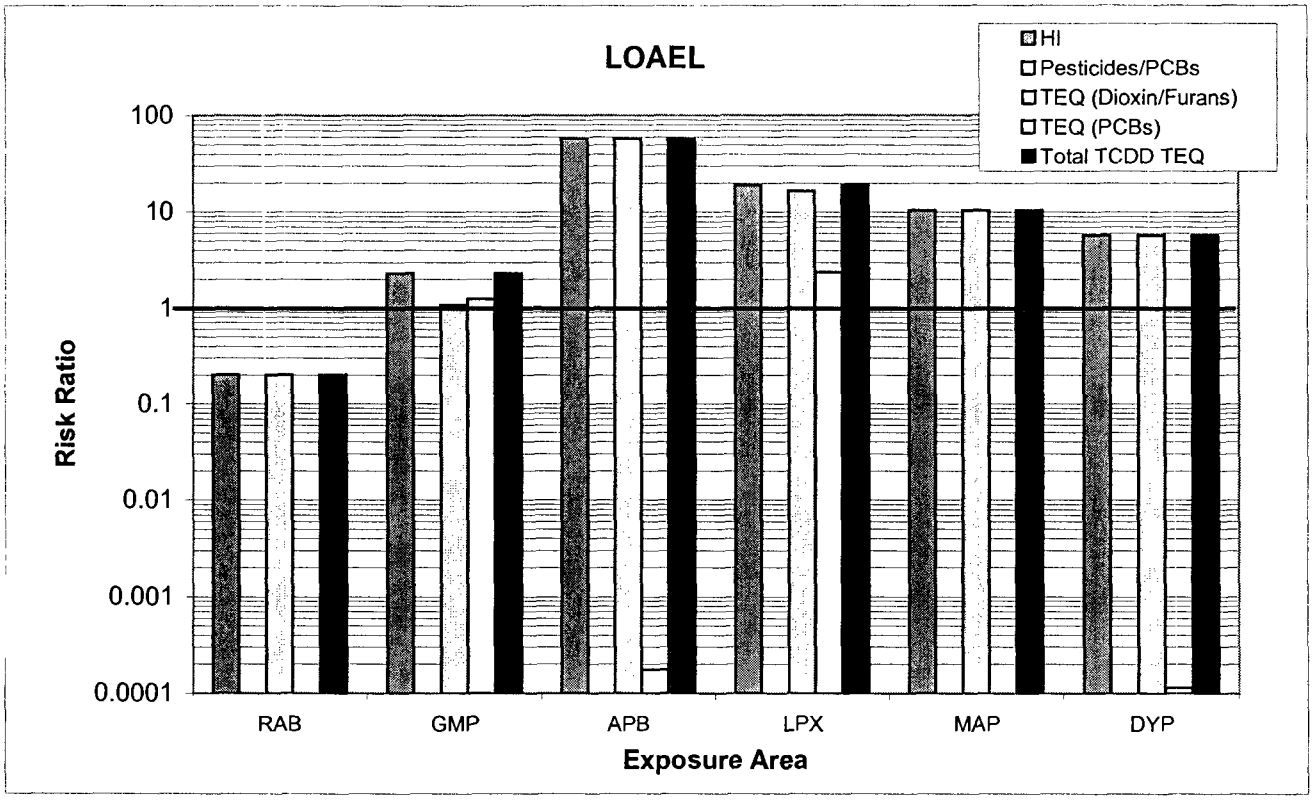


FIGURE 94
CBR RISK RATIOS - ESTIMATED LITTLE BROWN BAT TISSUE

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



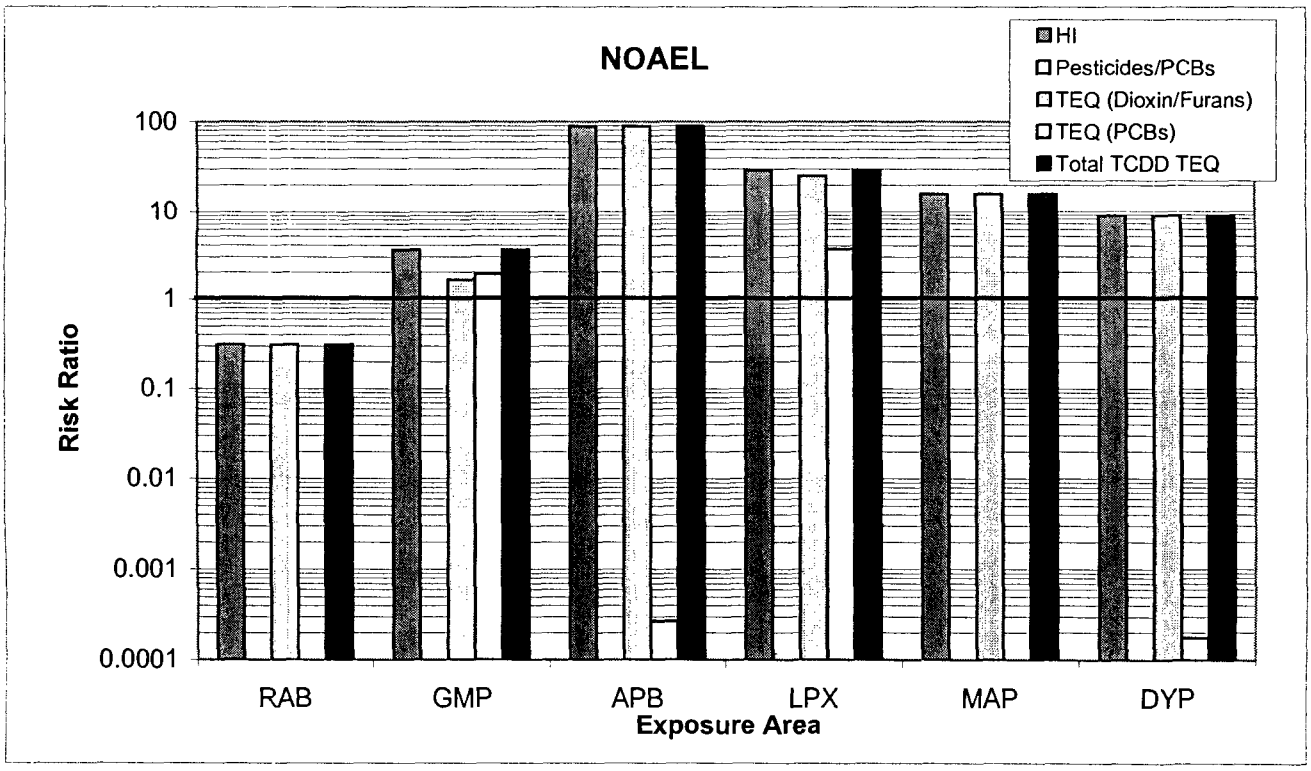
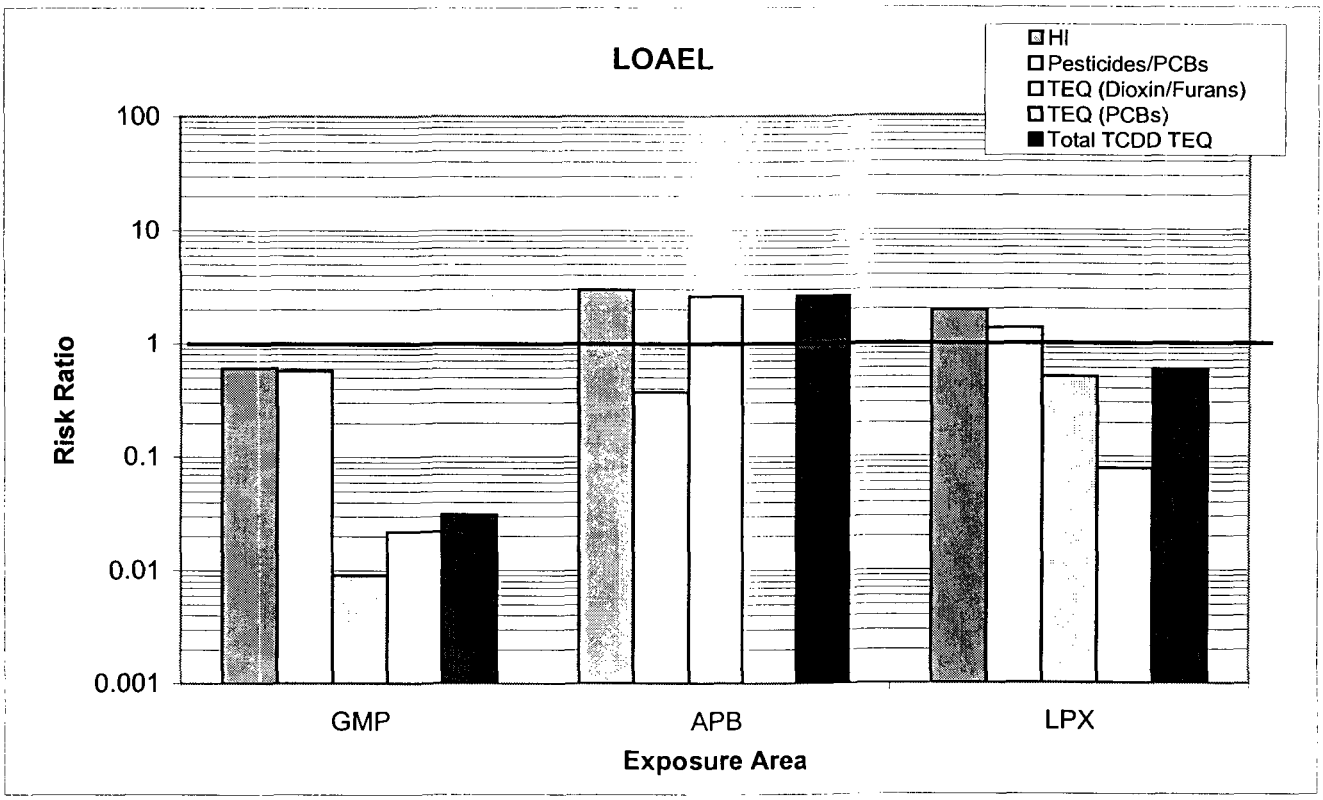
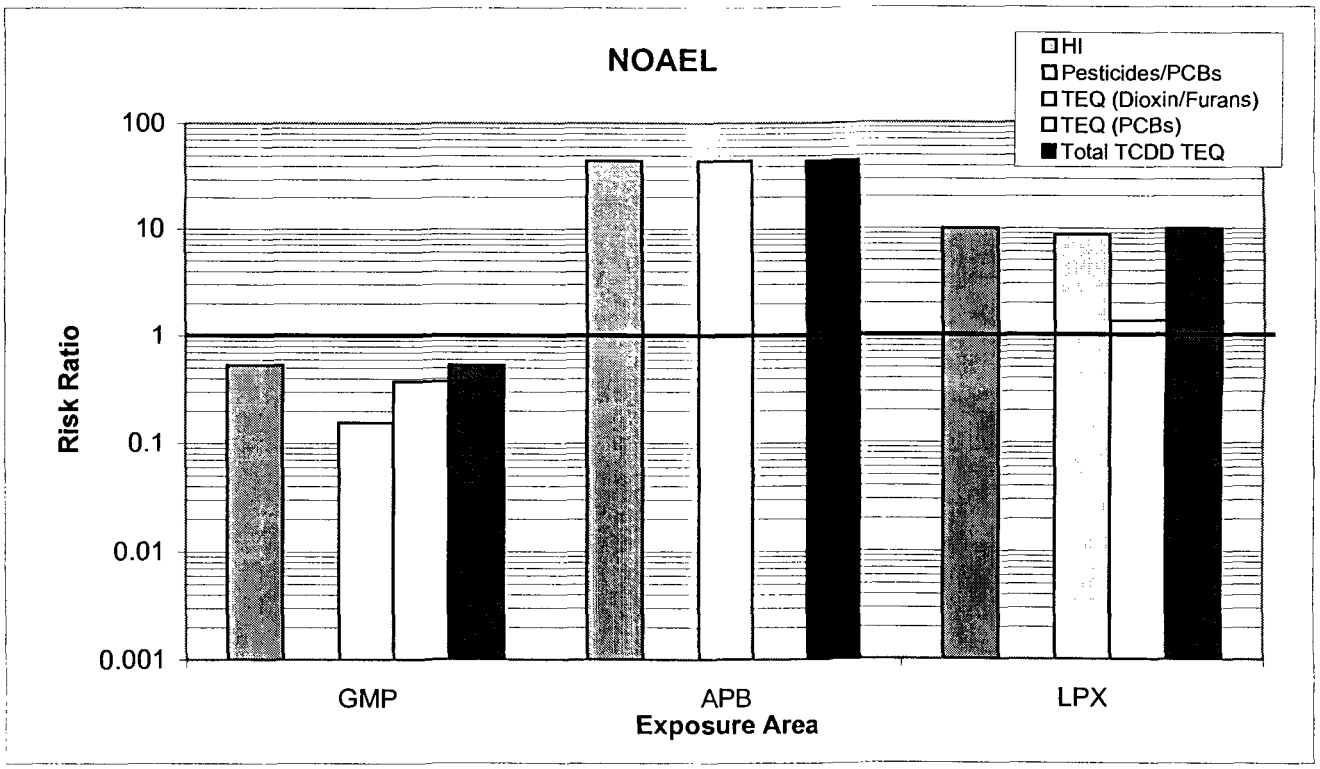


FIGURE 95
CBR RISK RATIOS -ESTIMATED VERMIVOROUS BIRD EGG TISSUE

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island





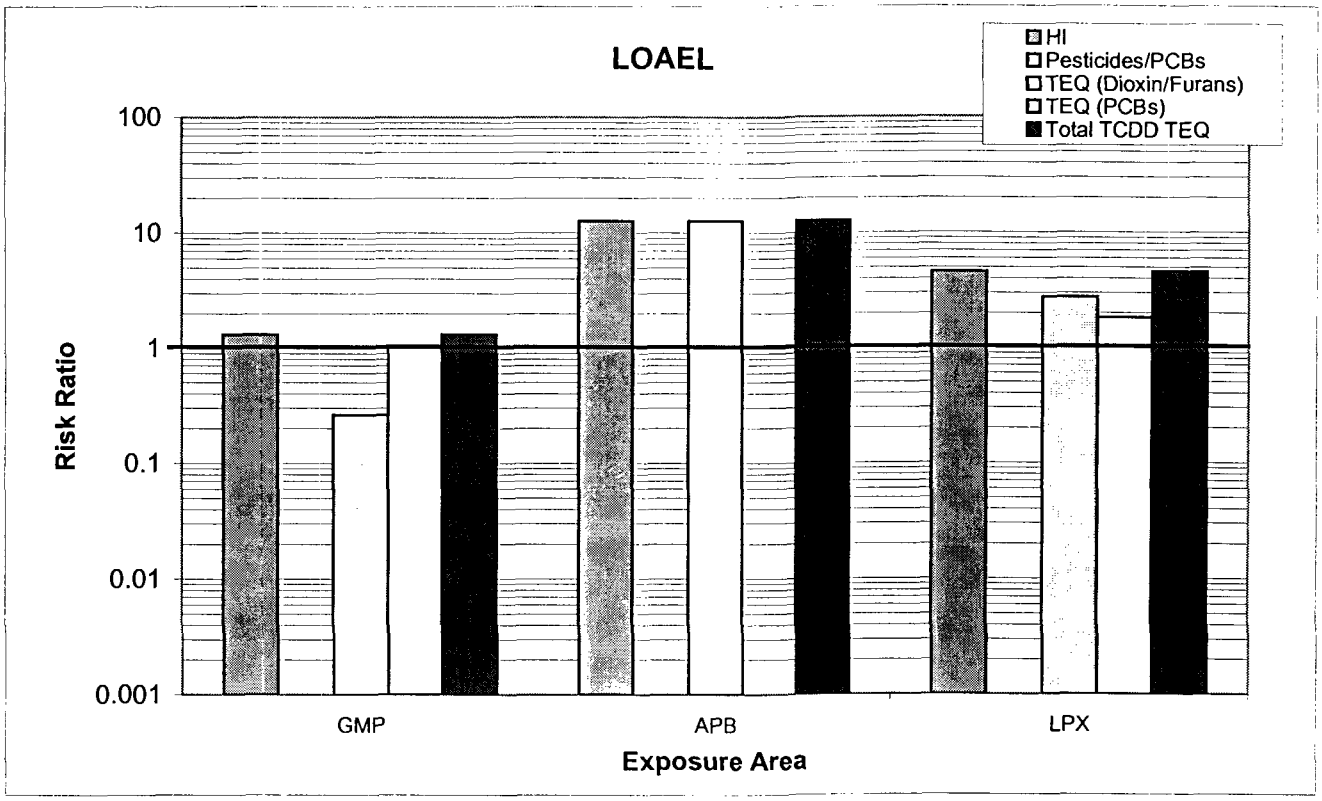
MACTEC Engineering and Consulting, Inc.
51226.24
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EW-BMF-TEFs(rev)Avian Egg

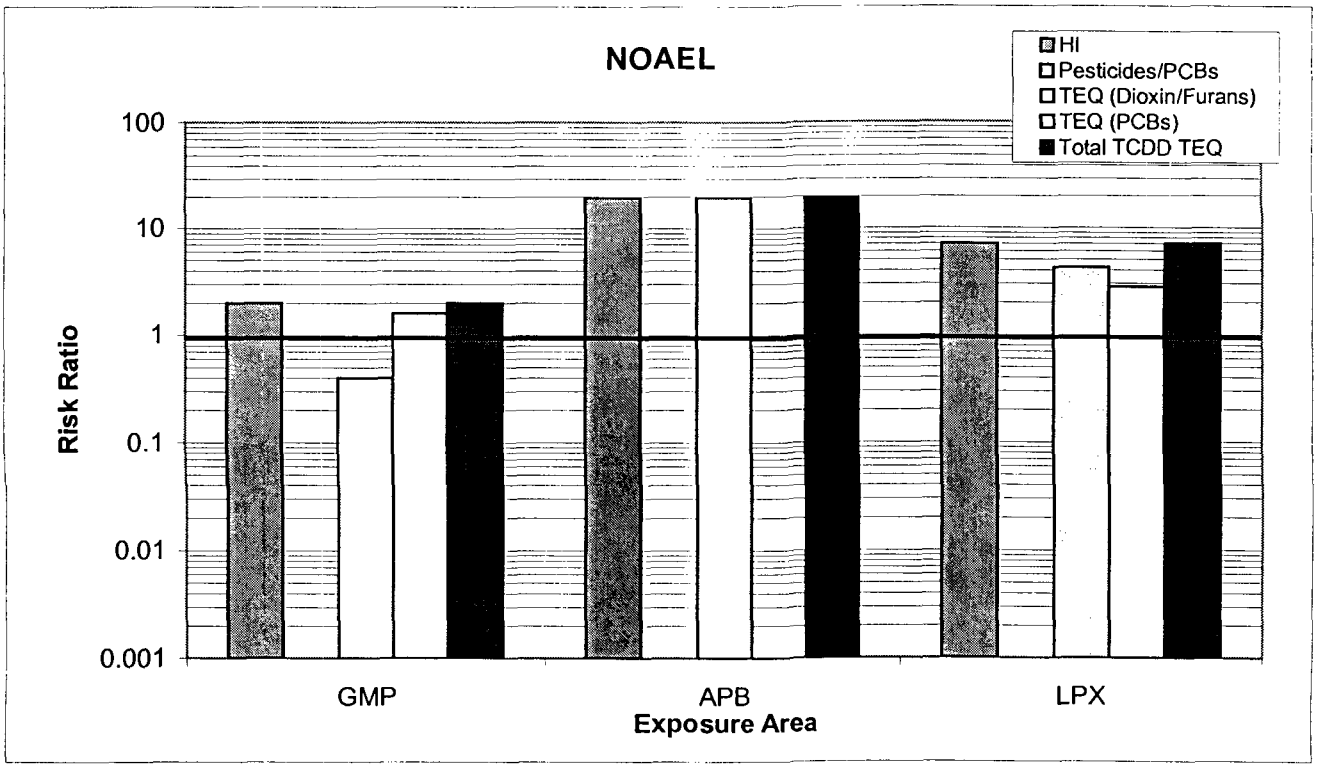
8/5/2004

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FIGURE 96
CBR RISK RATIOS - ESTIMATED VERMIVOROUS MAMMAL TISSUE

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



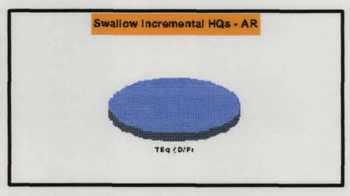


MACTEC Engineering and Consulting, Inc.
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EW-BMF-TEFs(rev)Mammal

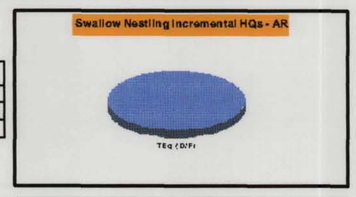
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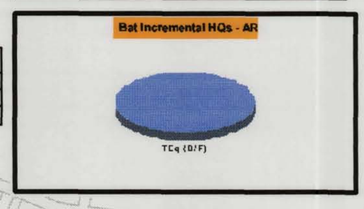
Incremental HQs - Swallow (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	2.9E+01	2.9E+00	100%
Incremental HI	**2.9E+01**	**2.9E+00**	



Incremental HQs - Swallow Nestling (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	2.1E+00	2.1E-01	100%
Incremental HI	**2.1E+00**	**2.1E-01**	



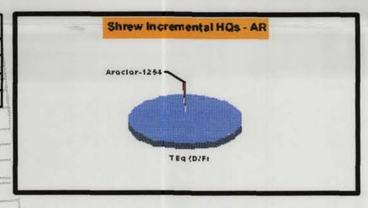
Incremental HQs - Bat (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	2.2E+02	2.2E+01	100%
Incremental HI	**2.2E+02**	**2.2E+01**	



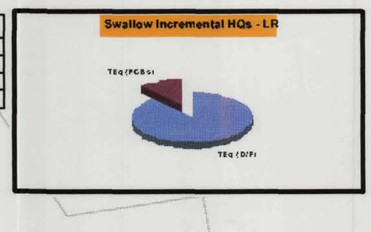
Incremental HQs - Woodcock (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	6.6E+00	6.6E-01	100%
Incremental HI	**6.6E+00**	**6.6E-01**	



Incremental HQs - Shrew (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	1.3E+02	1.3E+01	99%
PCB Aroclors	1.3E+00	1.3E-01	1%
Incremental HI	**1.3E+02**	**1.3E+01**	

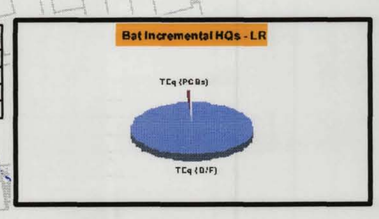


Incremental HQs - Swallow (LR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	7.6E+00	7.6E-01	88%
TEq (PCBs)	9.9E-01	9.9E-02	12%
Incremental HI	**8.6E+00**	**8.6E-01**	

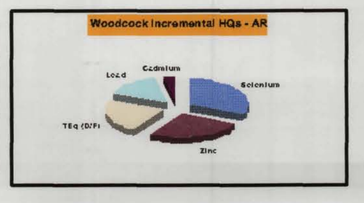


Incremental HQs - Swallow Nestling (LR)			
Analyte	NOAEL	LOAEL	%
None			

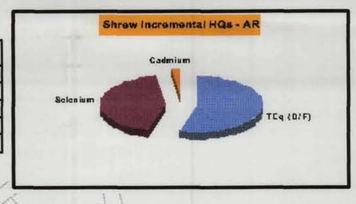
Incremental HQs - Bat (LR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	5.8E+01	5.8E+00	99%
TEq (PCBs)	4.4E-01	4.4E-02	1%
Incremental HI	**5.8E+01**	**5.8E+00**	



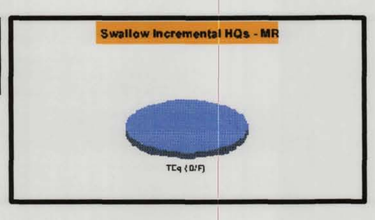
Incremental HQs - Woodcock (LR)			
Analyte	NOAEL	LOAEL	%
Selenium	9.8E-01	2.9E-01	32%
Zinc	2.2E+00	2.5E-01	27%
TEq (D/F)	2.0E+00	2.0E-01	22%
Lead	1.4E+00	1.4E-01	16%
Cadmium	4.1E-01	3.0E-02	3%
Incremental HI	**7.0E+00**	**9.1E-01**	



Incremental HQs - Shrew (LR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	3.4E+01	3.4E+00	54%
Selenium	4.5E+00	2.7E+00	43%
Cadmium	1.5E+00	1.5E-01	2%
Incremental HI	**4.0E+01**	**6.3E+00**	

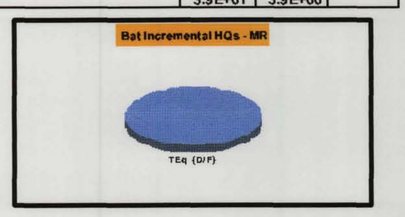


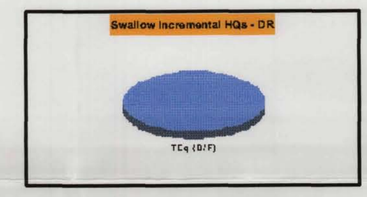
Incremental HQs - Swallow (MR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	5.0E+00	5.0E-01	100%
Incremental HI	**5.0E+00**	**5.0E-01**	



Incremental HQs - Swallow Nestling (MR)			
Analyte	NOAEL	LOAEL	%
None			

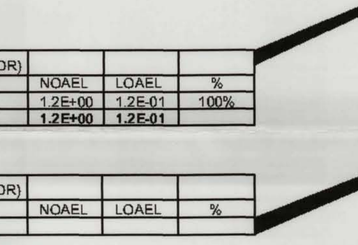
Incremental HQs - Bat (MR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	3.9E+01	3.9E+00	100%
Incremental HI	**3.9E+01**	**3.9E+00**	





Incremental HQs - Swallow (DR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	1.2E+00	1.2E-01	100%
Incremental HI	**1.2E+00**	**1.2E-01**	

Incremental HQs - Swallow Nestling (DR)			
Analyte	NOAEL	LOAEL	%
None			



Incremental HQs - Bat (DR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	7.9E+00	7.9E-01	100%
Incremental HI	**7.9E+00**	**7.9E-01**	

ALLENDALE REACH
ALLENDALE POND
ALLENDALE DAM
LYMAN MILL REACH
LYMAN MILL POND
LYMAN MILL DAM
MANTON REACH
MANTON POND
MANTON DAM
DYERVILLE REACH
DYERVILLE POND
DYERVILLE DAM
ASSAPUMPSET POND
ASSAPUMPSET BROOK
WOONASQUATUCKET RIVER
Flow Direction

NORTH

300 0 300 600
SCALE IN FEET

MACTEC
107 Audubon Road
Building II, #301
Wakefield, MA 01880
781-245-6606

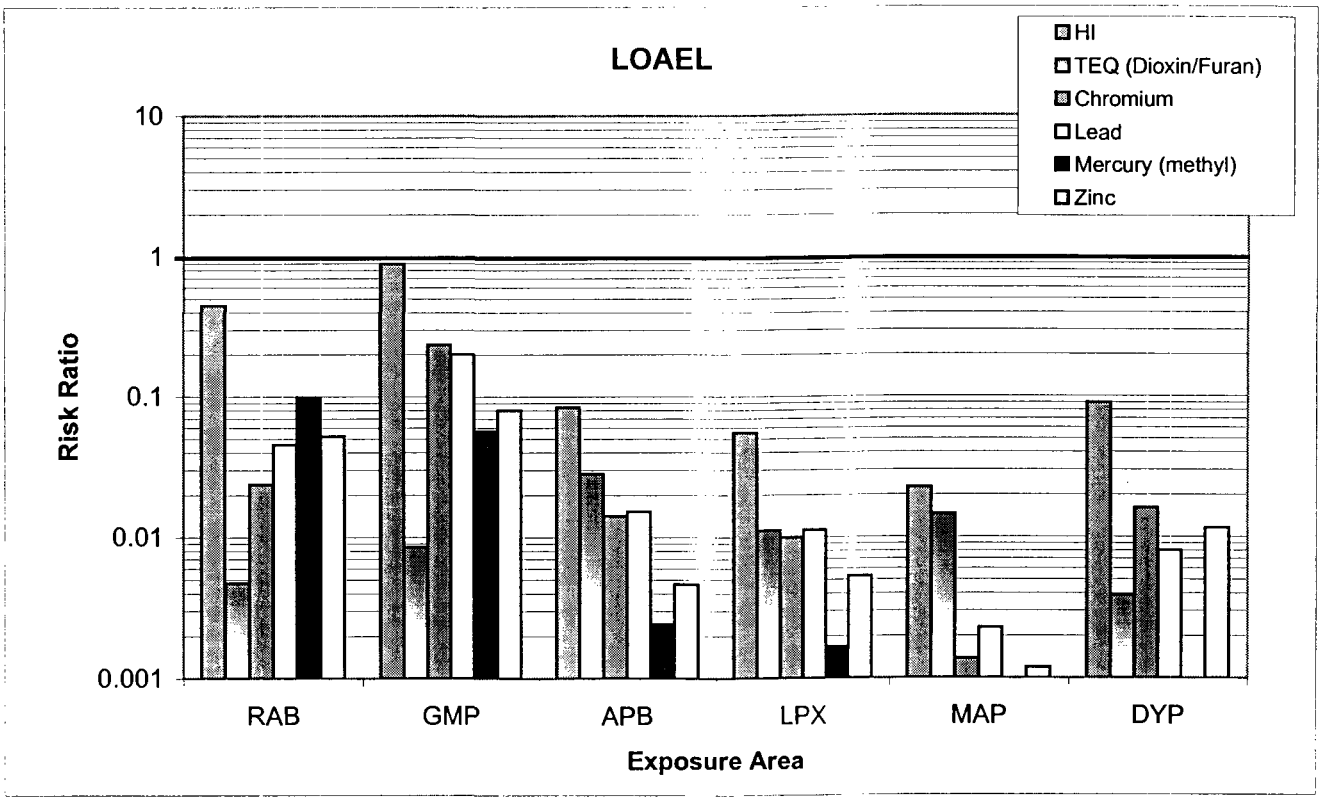
FIGURE 97
INCREMENTAL RISK SUMMARY FOR INSECTIVOROUS WILDLIFE
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

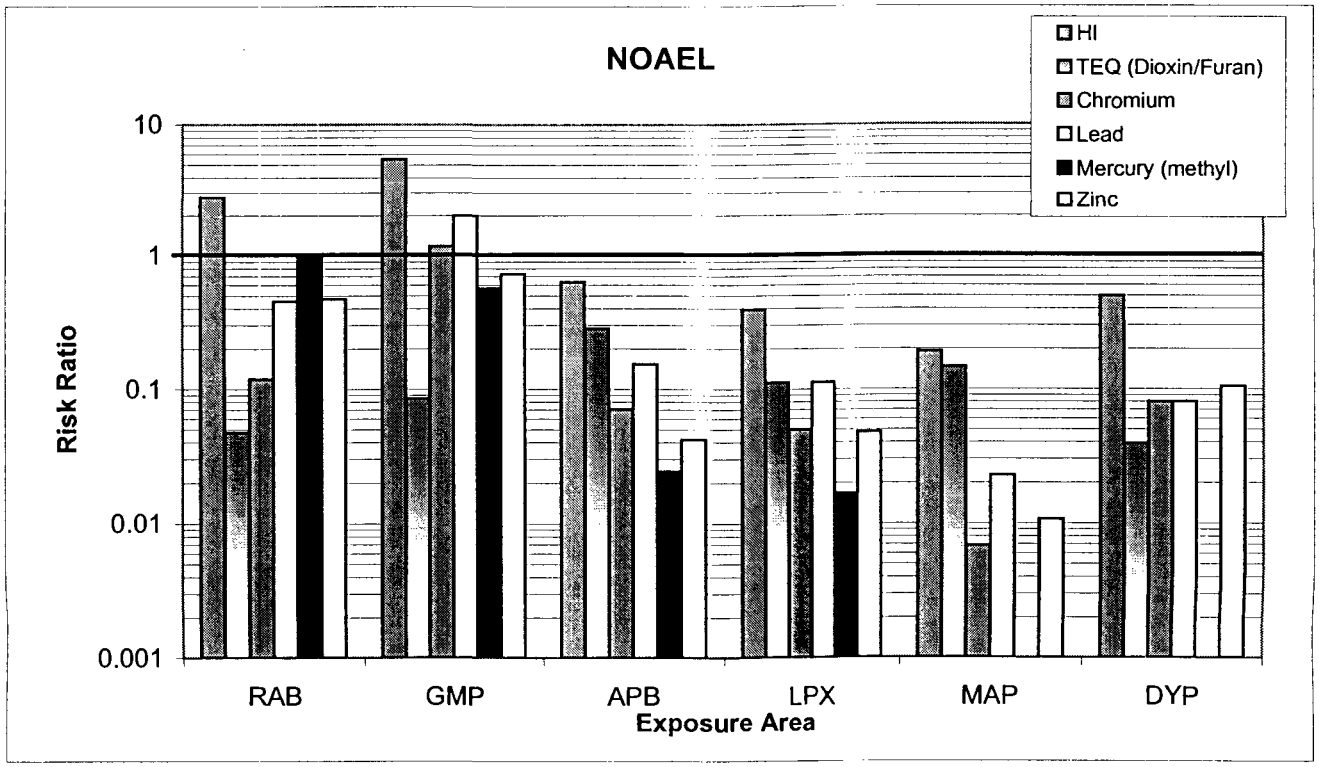
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centerdale2003c.dwg	BGF	51226.24	NAR	1/8/03	7/19/04

US EPA ARCHIVE DOCUMENT

FIGURE 98
SUMMARY OF WILDLIFE EXPOSURE MODELING - MALLARD

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

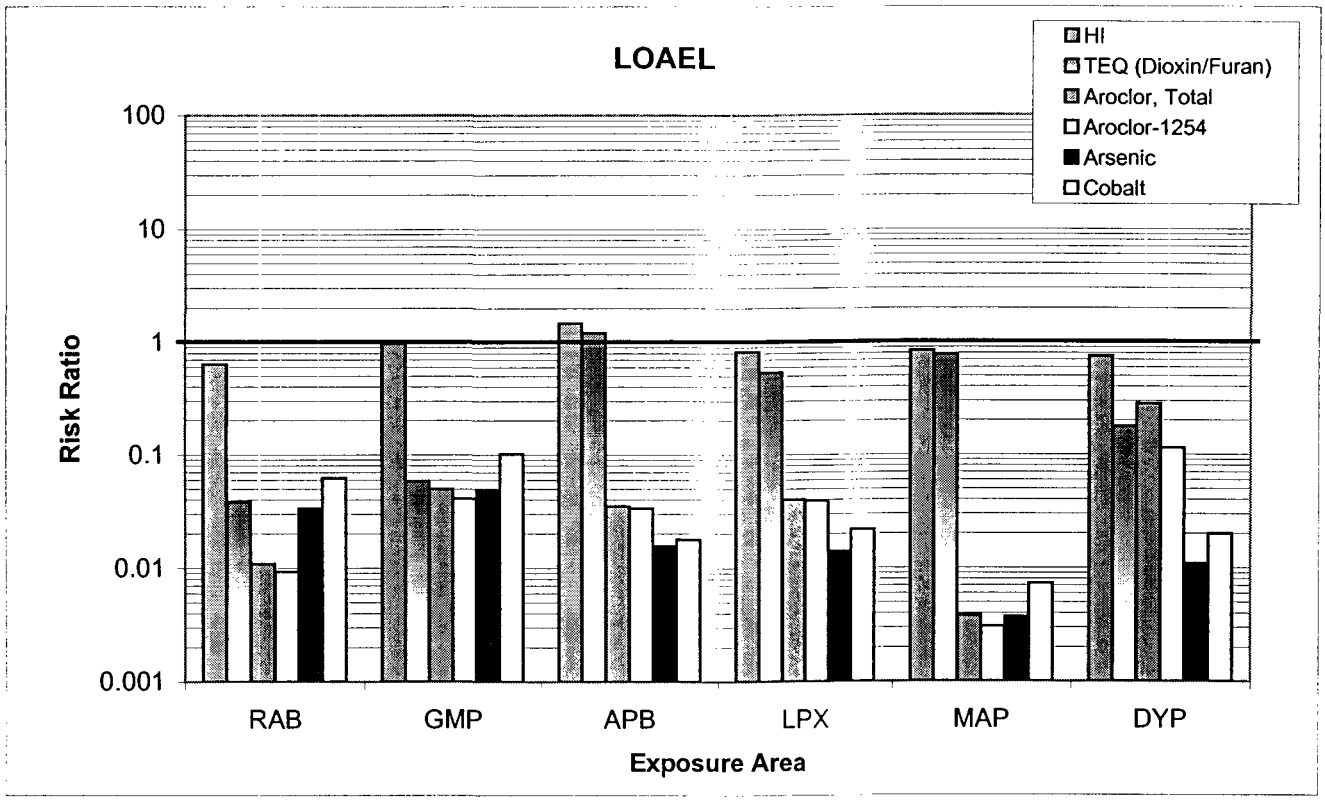


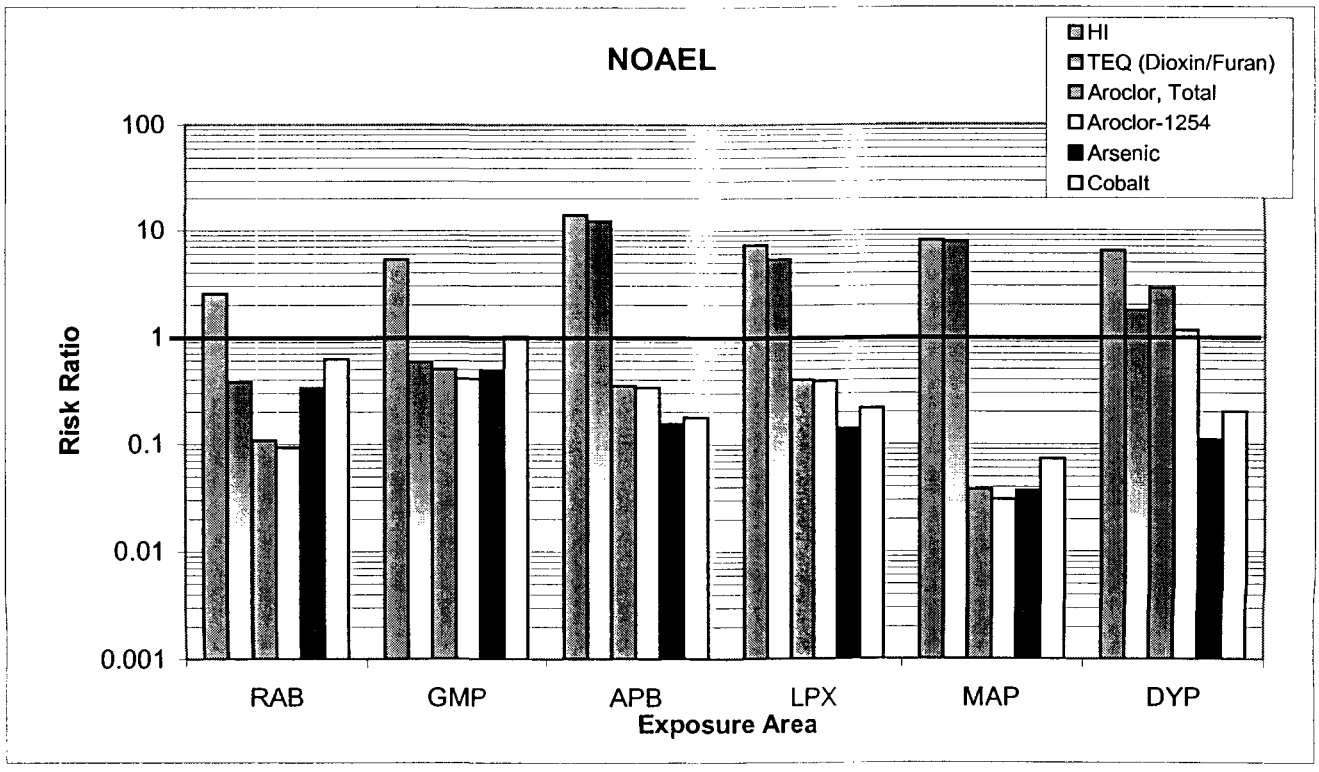


US EPA ARCHIVE DOCUMENT

FIGURE 99
SUMMARY OF WILDLIFE EXPOSURE MODELING - RACCOON (Aquatic Exposures)

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island

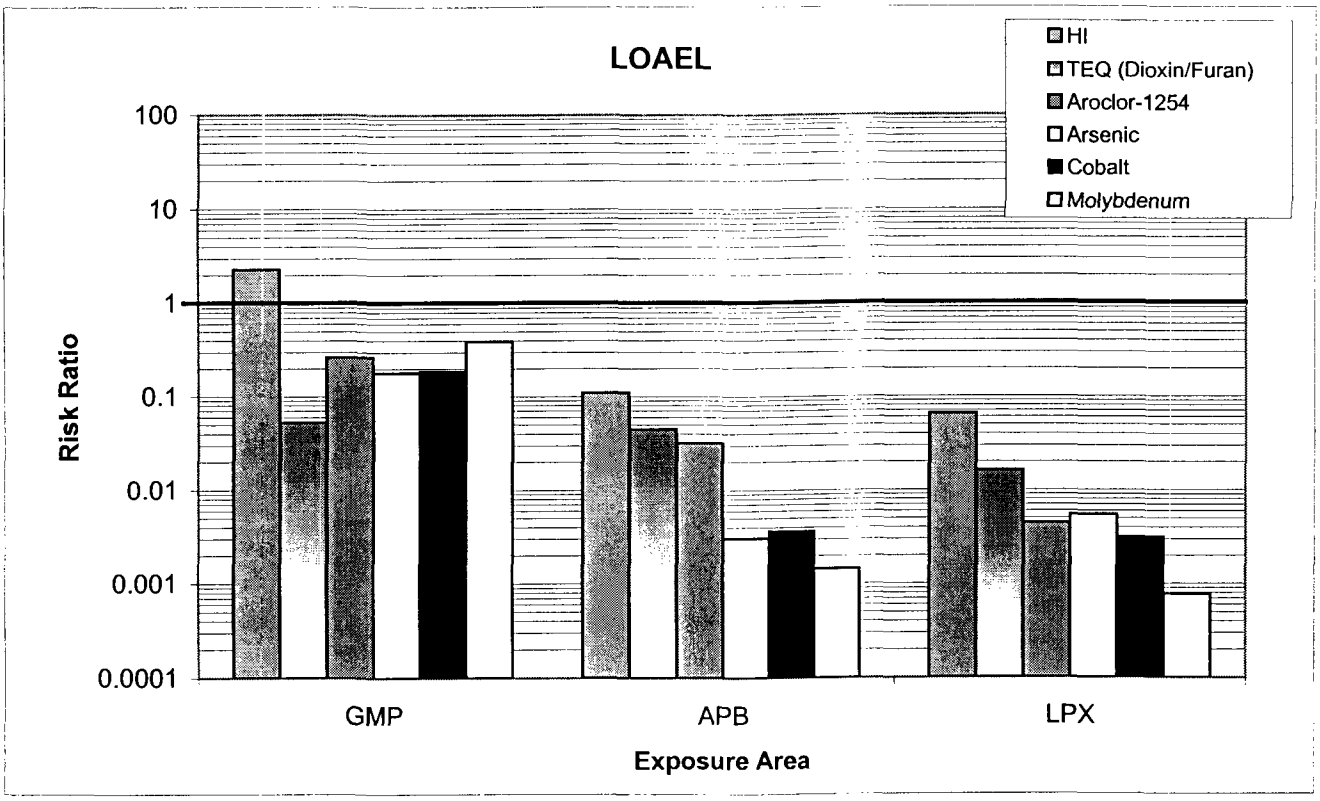


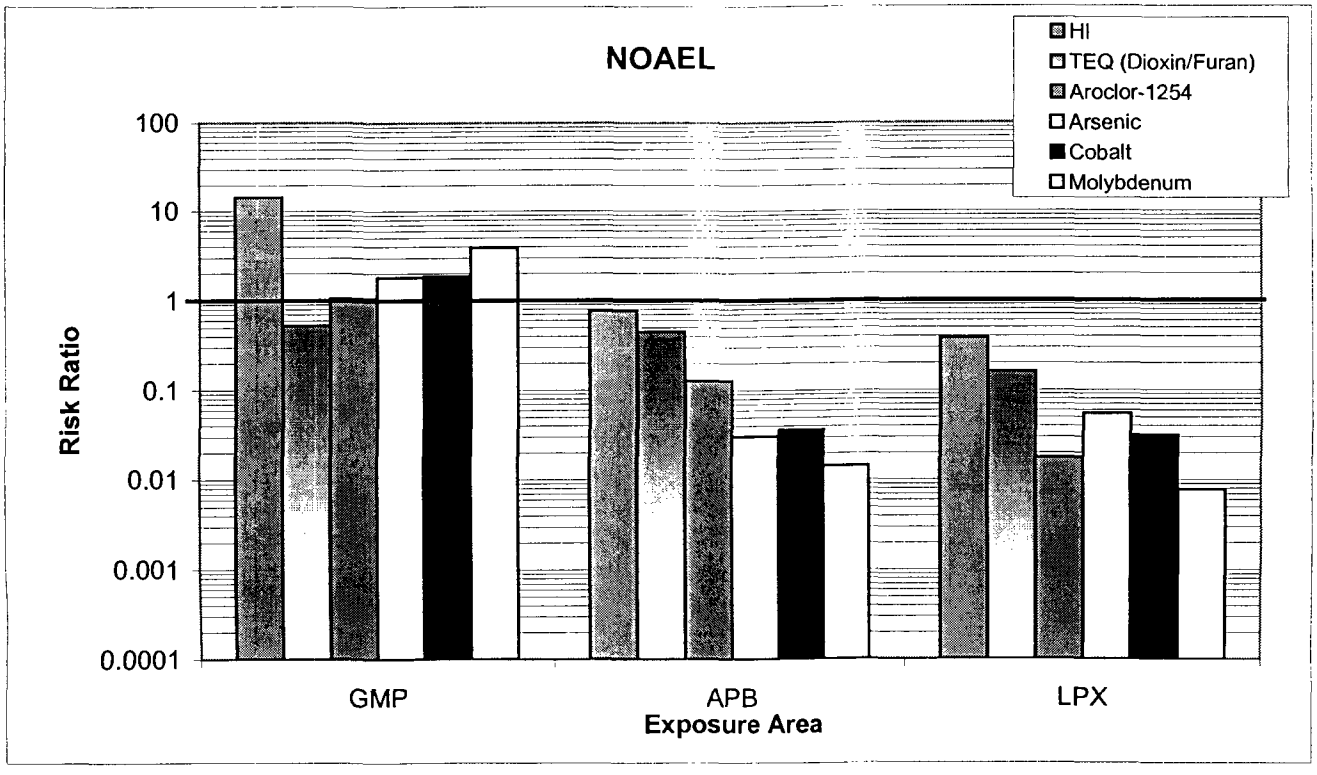


US EPA ARCHIVE DOCUMENT

FIGURE 100
SUMMARY OF WILDLIFE EXPOSURE MODELING - RACCOON (Floodplain Exposures)

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



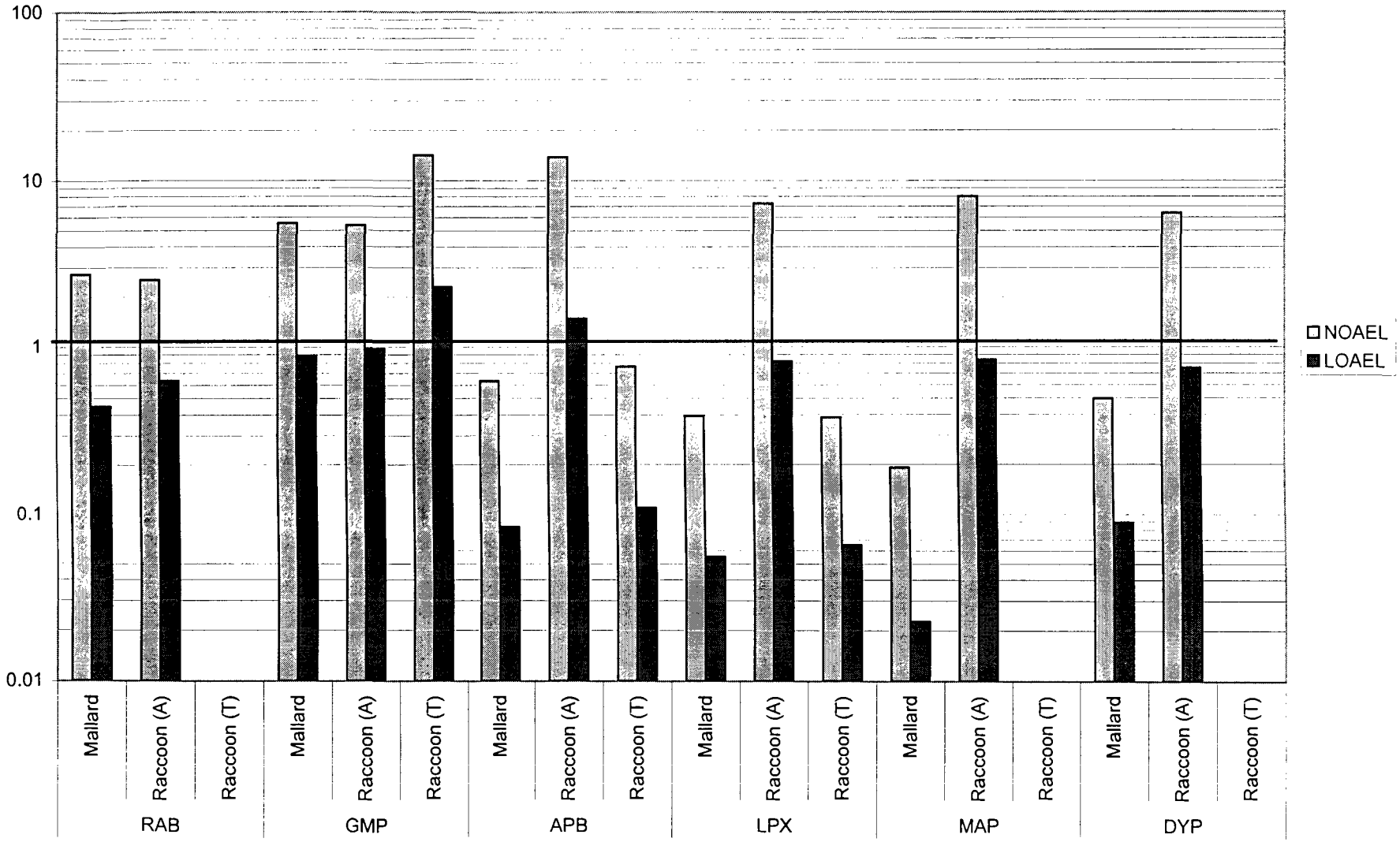


US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

FIGURE 101
FOOD WEB MODELING SUMMARY - OMNIVOROUS WILDLIFE RECEPTORS

Baseline Ecological Risk Assessment - Interim Final
Centredale Manor Restoration Project Superfund Site
North Providence, Rhode Island



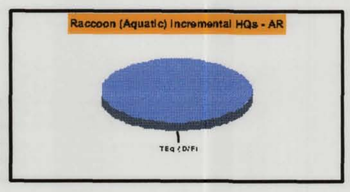
MACTEC Engineering and Consulting, Inc.
51226.24
P:\W9-GVT\COE-NAE\Battelle\Centredale\T24 - BERA\Interim Final\Appendices\KLN - Spreadsheets\
Omnivore Risk Summary(JUL04)Omnivore Risk Summary

8/5/2004

US EPA ARCHIVE DOCUMENT

Incremental HQs - Mallard (AR)			
Analyte	NOAEL	LOAEL	%
None			

Incremental HQs - Raccoon (Aquatic) (AR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	1.2E+01	1.2E+00	100%
Incremental HI	**1.2E+01**	**1.2E+00**	



Incremental HQs - Raccoon (Floodplain) (AR)			
Analyte	NOAEL	LOAEL	%
None			

ALLENDALE POND

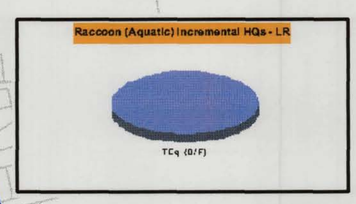
ALLENDALE DAM

ALLENDALE REACH

Incremental HQs - Mallard (LR)			
Analyte	NOAEL	LOAEL	%
None			

LYMAN MILL REACH

Incremental HQs - Raccoon (Aquatic) (LR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	4.8E+00	4.8E-01	100%
Incremental HI	**4.8E+00**	**4.8E-01**	



ASSAPUMPSET POND

ASSAPUMPSET BROOK

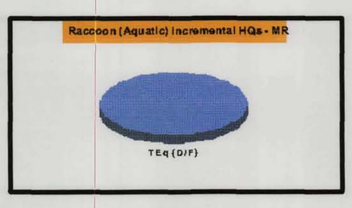
WOONASQUATUCKET RIVER

LYMAN MILL POND

LYMAN MILL DAM

Incremental HQs - Mallard (MR)			
Analyte	NOAEL	LOAEL	%
None			

Incremental HQs - Raccoon (Aquatic) (MR)			
Analyte	NOAEL	LOAEL	%
TEq (D/F)	7.2E+00	7.2E-01	100%
Incremental HI	**7.2E+00**	**7.2E-01**	

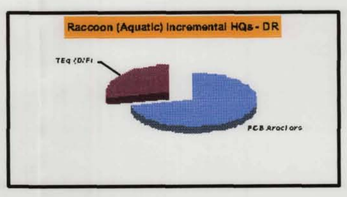


MANTON REACH

MANTON POND

MANTON DAM

Incremental HQs - Mallard (DR)			
Analyte	NOAEL	LOAEL	%
None			



Incremental HQs - Raccoon (Aquatic) (DR)			
Analyte	NOAEL	LOAEL	%
PCB Aroclors	3.0E+00	3.0E-01	72%
TEq (D/F)	1.2E+00	1.2E-01	28%
Incremental HI	**4.2E+00**	**4.2E-01**	

DYERVILLE REACH

DYERVILLE POND

DYERVILLE DAM

300 0 300 600

SCALE IN FEET

MACTEC
107 Audubon Road
Building II, #301
Wakefield, MA 01880
781-245-6606

FIGURE 102
INCREMENTAL RISK SUMMARY FOR OMNIVOROUS WILDLIFE
CENTREDALE MANOR RESTORATION PROJECT SUPERFUND SITE
NORTH PROVIDENCE, RHODE ISLAND

FILE NUMBER:	DRAWN:	JOB NUMBER:	APPROVED:	DATE:	REVISED DATE:
centerdale2003d.dwg	BGF	51226.24	NAR	1/8/03	7/19/04